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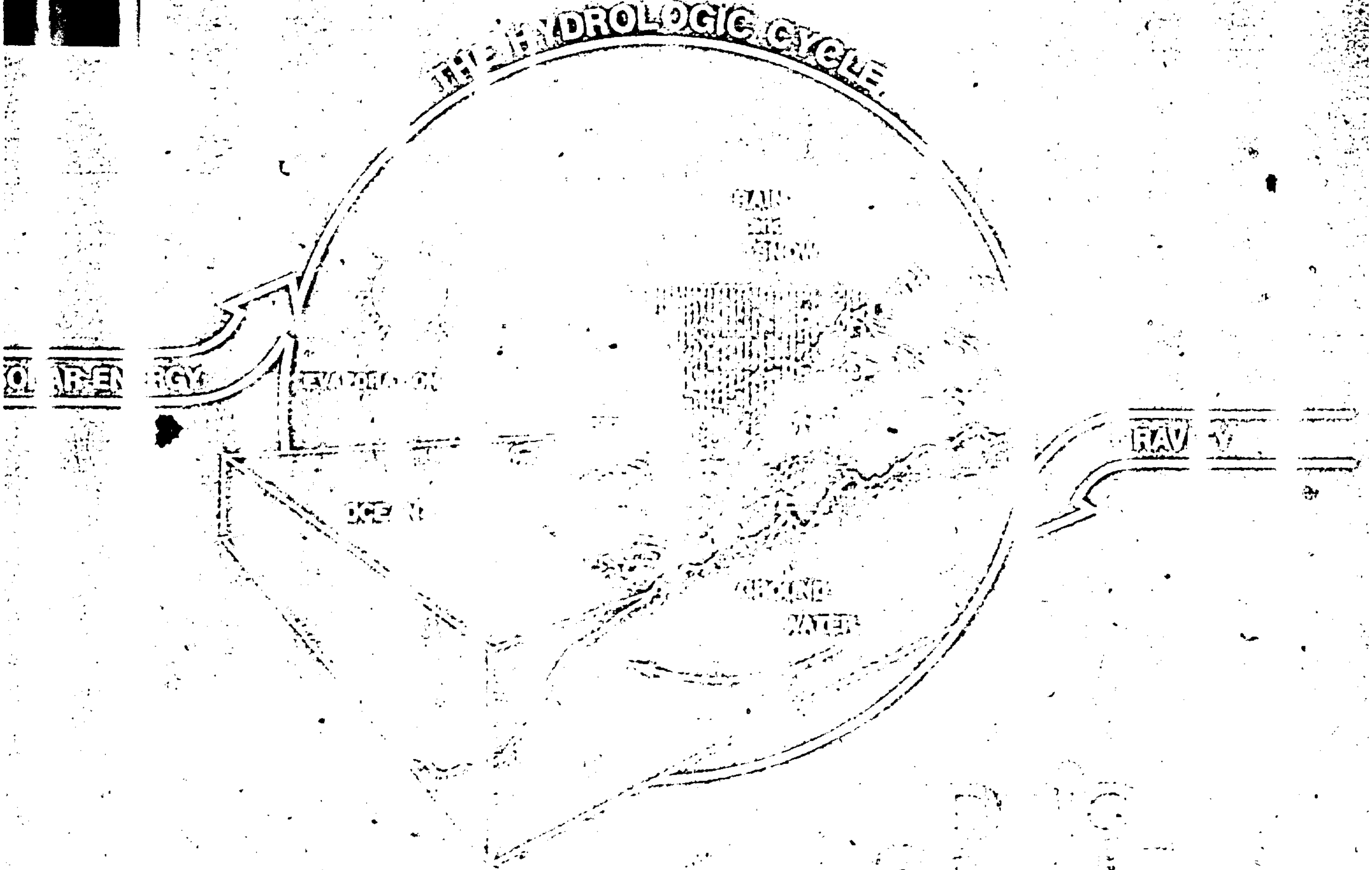
PROCEEDINGS: NATIONAL WETLAND SYMPOSIUM

WETLAND HYDROLOGY



September 16-18, 1987
Chicago, Illinois

Association of State Wetland Managers, Inc

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Wetland Hydrology

September 16–18, 1987
Chicago, Illinois

Symposium Chairman: Jon A. Kusler

Co-Chairmen: William Niering
Richard Novitzki

Editors: Jon A. Kusler
Gail Brooks

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FOREWORD

This collection of papers is designed to help wetland managers (regulators, planners, researchers, waterfowl managers) understand wetland hydrology, its relationship to various wetland functions, the impact of various activities on hydrology, and approaches for reducing or compensating for those impacts. It is the first attempt to collect, in one volume, papers representing a broad range of disciplines and perspectives concerning wetland hydrology. This volume has been prepared because the lack of literature and the inaccessibility of many of the papers and reports that address this subject has been one of the problems encountered by the wetland manager in addressing wetland hydrology.

The papers are drawn primarily from a three-day symposium on wetland hydrology conducted by The Association of State Wetland Managers in Chicago in September, 1987. A brief primer on wetland hydrology and several papers concerning hydrology from earlier proceedings conducted by the Association are also included.

Wetland hydrology is a complicated and many-faceted subject. We know that this volume will, by no means, be the last word on the subject, but we hope that it will help stimulate wetland research, training of wetland managers in hydrology and better exchange of information between wetland biologists, hydrologists, geomorphologists, engineers and others. We hope that the papers will help you better address issues or lead you to other useful sources of information.

ACKNOWLEDGEMENTS

This report would not have been possible without the help of the co-chairmen: Professor William Niering of Connecticut College and Dr. Richard Novitzki of the U.S. Geological Survey. Their assistance in all phases is much appreciated.

The report could not have been prepared without the many volunteers who participated, as well as the sponsorship of the Illinois Department of Conservation, the U.S. Environmental Protection Agency, and the U.S. Army Corps of Engineers. Particular thanks are due Marvin Hubbell and the staff of the Illinois Department of Conservation and Doug Ehorn of the Chicago EPA office and his staff for helping to design the program and to carry out the actual symposium. Special appreciation is due Eric Preston and Mary Kentula of the EPA Corvallis Laboratory for helping to fund these proceedings. Additional assistance was provided by Colonel Kit Valentine and his staff at the U.S. Army Corps of Engineers.

Dr Donald Hey and his staff at Wetland Research, Inc. designed and conducted an excellent field trip for symposium participants to Wetland Research's wetland restoration site on the Des Plaines River.

Gail Brooks, our editor and publications coordinator, provided excellent editing, typesetting, and layout assistant. Thanks Gail for your patience and persistence!

Finally, thanks are due our sponsoring and cooperating parties and the many excellent speakers and participants.

Sincerely,

Jon Kusler
Symposium Chairman

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RECOMMENDATIONS: STRATEGIES FOR ADDRESSING HYDROLOGIC ISSUES

Assuming limited financial resources, manpower, and hydrologic expertise, how can a wetland manager best approach hydrologic issues in reviewing a permit application, preparing or reviewing a plan for wetland restoration, preparing an E.I.S., or carrying out other management activities?

Several general strategies can be suggested:

--Presume that all hydrologic parameters (depth, velocity, etc.) in a naturally occurring wetland are important to specific functions and the long term continued existence of the wetland. The burden of overcoming this presumption should be shifted to those proposing changes in wetland hydrology. Any major changes in wetland hydrology should be approached with care. While changes may not always be destructive, a presumption that all parameters are important is justified. All wetland functions, not just the traditional wetland functions such as flood storage and groundwater recharge, depend upon the hydrologic regime.

--Assess the role of not only "mean" or "normal" hydrologic conditions but extreme events and long term fluctuations in permitting or planning activities which impact upon wetland hydrology. Consideration of extreme events and long term fluctuation is particularly important for wetland restoration, enhancement, or creation.

--Apply what is known about hydrology in wetland decision-making and approach areas of uncertainty conservatively. Apply "safety factors" much like engineers apply such factors to design of structures. Although much is not understood from a basic scientific perspective about wetland hydrology, much more is known than is being applied. The rationale that "much is unknown" should also not be used as an excuse for ignoring wetland hydrology or failing to apply what is known.

Additionally, more specific strategies for the wetland manager may include the following:

General Approaches

--When faced with a permit application or planning task involving changes in hydrology, restoration, creation, or enhancement, begin by defining the "right questions" for permittees, your own staff, or consultants. Define "critical" hydrologic parameters (e.g., depth, velocity, hydroperiod) early so that you can establish priorities for data-gathering and data analysis.

--Recognize that the hydrology of a specific wetland cannot, in most instances, be described or analyzed in terms of a single set of conditions since water depths, velocities, sediment loadings, and other hydrologic characteristics depend directly or indirectly upon long and short term variations in precipitation and tides (coastal and estuarine wetlands). Relevant conditions may include mean annual conditions, and annual and long term maxima and minima.

--Recognize that wetland hydrology, including the shape and form of a wetland, is dynamic and will change due, in part, to the impacts of hydrologic events including sediment loadings and erosion, and also to processes within the wetland including accumulation of organic material and the work of micro and macroorganisms (e.g., beaver). This is particularly true for wetlands in the channels of or adjacent to high gradient streams and other wetlands located in areas of high energy.

--Seek the assistance of an interactive team and work in this context with hydrologic and geomorphological experts (if you are not one).

Presumptions

--Presume that any new "hydrologic" expert you employ will need help in defining the long and short term hydrologic and geomorphologic parameters needing study in a particular instance. Few hydrologic experts understand the relationship of hydrology to specific wetland functions and few have expertise in all aspects of wetland systems--groundwater and surface water, normal flows and flood flows, surface water and water/sediment relationships.

--Presume that the wetland is hydrologically and geomorphologically part of the adjacent lake, ocean, stream (if one exists) for management purposes. Wetlands are rarely hydrologically or geomorphologically separated from adjacent water bodies and cannot be managed separately.

--Presume that efforts to stabilize hydrologic parameters and reduce fluctuations and extreme events may bring short-term increases in certain functions but that variations in hydrologic parameters within a certain range may be essential to the long-term continued existence of the wetland. Hydrologic maxima and minima, including extreme flood events, often interrupt

traditional successional sequences, contributing to the long term maintenance of a wetland and wetland functions.

--Presume that, for most functions and values, the following hydrologic characteristics are important:

- sources of water (surface, ground, direct precipitation) and pathways into, through, and out of the wetland (transpiration, evapotranspiration, groundwater recharge, surface flow),
- depth of water (flood conveyance, fishery potential, recreation potential, vegetation type, etc.),
- velocity of water (erosion potential, pollution control potential, etc.),
- dissolved and suspended substances (including sediment) in the inflowing and outflowing water,
- shape of the wetland including configuration of the bank (flood storage potential, flood conveyance potential, groundwater recharge potential, etc.),
- connection to ground or surface waters (fishery potential, pollution control potential, flood storage and conveyance potential, food chain support, etc.)
- soil substrate (pollution control potential, vegetation, erosion potential), and
- vegetation and animal life (flood conveyance, pollution control potential, habitat value, food chain

Assessment Strategies

--Assess the possible impacts of hydrologic changes upon all wetland values, not simply "hydrologic functions", since all functions depend directly or indirectly upon hydrology.

--To reduce costs of hydrologic studies, utilize existing data sources and what can be gained from brief field surveys as much as possible. Use flood maps to help project the role of extreme events. Where more detailed modeling and/or long term studies and monitoring are needed, cost-saving may be realized by careful design and selective siting of instruments.

--Recognize that a complete understanding of wetland hydrology is often not necessary (as well as impossible) and the information needed will depend upon the nature of the proposed activity (e.g., a drainage ditch versus a fill), the nature of the wetland, and the likely functions which may be impacted.

--Recognize that there are margins of error and limits to precision in hydrologic and hydraulic studies. Do not waste time and resources striving for accuracy in one calculation where margins of error in another render such precision of limited value. Require hydrologic experts to articulate probable margins of error, limits to precision, and ranges in their measurements.

Management Strategies

--Wherever changes are proposed in a naturally occurring wetland, attempt, as a first priority, to protect the hydrologic regime. Vegetation may be cut, soils may be disturbed, but as long as the natural hydrologic regime is intact, wetland functions often can be restored or may restore themselves naturally.

--Emphasize flood loss reduction, prevention of pollution, protection of water supply and protection of other wetland hydrologic functions and values in carrying out regulatory analyses, particularly where denial of a permit may "take" private property. Courts have given particular weight to efforts to reduce flood losses, protect water supply, reduce pollution, protect groundwater discharge, reduce erosion, and serve other "hydrologically" related objectives and almost never find that regulatory restraints designed to serve these objectives "take" property.

--Where water levels or other hydrologic parameters are to be actively managed over time, insure that the institutional mechanisms, motivation, funding, and levels of expertise are sufficient for such manipulation. Consider the possible adverse impacts of stabilization of water levels.

--Encourage property-specific, community, and regional wetland management planning for watersheds or sub-watersheds, based upon hydrologic analyses for the entire water systems and designed to accomplish multiobjective goals (wetland protection and restoration, stormwater management, flood control, water quality protection, navigation).

chapter one

Introduction

Hydrology:
An Introduction for Wetland Managers

Jon Kusler
Association of State Wetland Managers

INTRODUCTION

Wetland managers (planners, regulators, consultants) recognize that some understanding of hydrology is necessary to prepare protection and management plans for particular wetlands, to analyze permits, to design impact reduction measures, to undertake active management activities (e.g., water level manipulation) for waterfowl or other purposes, and to create or restore wetlands. But it is one matter to recognize the importance of hydrology and another to assess and understand complex wetland/hydrologic relationships.

Wetland hydrology may be broadly defined to include the flow of water (precipitation, ground water, surface water) into, through, and out of a wetland, the characteristics of this flow, and its interaction with the wetland. In the short term, hydrology determines vegetation, fauna, and most wetland functions. In the longer term, hydrology determines, through erosion and deposition, the shape, size, depth, and even the location of a wetland. This, in turn, determines vegetation, fauna, and wetland functions.

Wetland managers have usually focused upon short term hydrologic relationships. But, the influence of hydrology upon the basic shape, size, and depth (overall form) of the wetland is equally important, in some contexts, if the goal of wetland management is to protect and manage wetland functions in terms of many decades and hundreds of years. The critical short term hydrologic parameters are often water depth and dissolved and suspended materials. The critical long term hydrologic parameters include water velocity and sediment loadings in addition to the short term parameters.

To properly evaluate proposed changes in hydrology or to prepare plans for wetland restoration, enhancement, or creation, wetland managers need to know how hydrologic maxima, minima, and "mean" conditions determine particular characteristics and functions. They need to know how specific activities impact wetland hydrology and how these changes will directly or indirectly affect wetland characteristics and functions. They need to know how wetlands will respond to hydrologic maxima and minima and to various intermediate conditions. They need to know the degree of sensitivity of wetlands to changes in hydrology and whether impacts will be long- or short-term. In evaluating mitigation proposals, they need to know what hydrologic conditions are needed to restore a wetland or create a new one with particular functions and values. They need to know how to deal with hydrologic issues in situations in which data is limited and wide ranges exist for possible error in calculations.

An understanding of wetland hydrology was perhaps less important a decade ago when regulatory and other management efforts focused upon total protection of naturally occurring wetlands. As wetland assessment and management efforts have become more sophisticated and active and wetland restoration and creation efforts more common, the need to understand wetland hydrology has increased dramatically.

But, even now wetland managers often assess and manage wetlands as isolated "islands" of hydrophytic vegetation and soils with their "wetness" assumed. The source of wetland water is not considered nor the relationship of the wetland to upland (watershed) water sources or adjacent deep water. Most wetland studies conducted by wetland managers are short-term and continue to focus upon wetland plants and animals.

One reason for the failure of wetland managers to consider hydrology is that wetland regulatory and other management programs are often separated at federal, state, and local governmental levels from floodplain, public water, lake, coastal and river management programs. Experts in hydrology and engineering are typically hired for floodplain and other "water" programs, not the wetland programs. It is sometimes assumed that such programs adequately deal with wetland hydrologic issues such as protecting minimum required flows in streams, but this assumption is usually wrong.

In a more basic sense, wetland managers have difficulty assessing wetland hydrology because calculations pertaining to the water entering and leaving a wetland and the interaction of this water with the vegetation, fauna, and soils of the wetland often require

unfamiliar data gathering (e.g., flow gauging) and a good deal of unfamiliar modeling and mathematics. Hydrologic and hydraulic studies can also be time consuming, difficult, and expensive. Hydrology is an increasingly complicated subject. Some of the reluctance of managers to address hydrologic issues may also be due to contradictory and apparently confusing information provided by hydrologists themselves who often know a great deal about specific hydrologic phenomena but nothing about the hydrologic requirements of specific wetland vegetation and fauna nor the interactions between the wetlands and hydrologic events.

The following paper provides a brief introduction to wetland hydrology which may be of some value to wetland managers beginning to grapple with hydrologic issues. It addresses common issues and, in some instances, common misconceptions concerning hydrology. It begins with a general discussion of issues and concludes with more specific discussion concerning wetland/groundwater and wetland flood loss reduction relationships. The paper is based upon the papers contained in this volume, discussion at the workshops, additional discussion with workshop speakers, and the author's own experiences over the last twenty years working with hydrologists and geomorphologists as well as more traditional wetland managers.* The paper should be read in conjunction with the introductory recommendations to this report: Recommendations; Strategies for Addressing Wetland Hydrology. The author apologizes for any over-simplifications of a very complex subject.

1. IN WHAT CIRCUMSTANCES DOES A WETLAND MANAGER NEED TO KNOW ABOUT WETLAND HYDROLOGY ?

Some understanding of wetland hydrology is desirable to carry out a broad range of routine wetland management activities. It is essential where activities are proposed which may alter natural hydrology or where a wetland is to be restored or created.

Some understanding of wetland hydrology is important to an assessment of virtually all wetland functions and the impact of various activities on those functions. As Mitch and Gosselink in their recent wetland text (1986) state: *"(h)ydrology is probably the single most important determinant for the establishment and maintenance of specific types of wetlands and wetland processes"* (emphasis added by the authors). They further note that "(w)hen hydrologic conditions in wetlands change even slightly, the biota may respond with massive changes in species richness and ecosystem productivity. When the hydrologic pattern remains similar from year to year, a wetland's structural and functional integrity may persist for many years."

Knowledge of wetland hydrology is critical to wetland creation efforts. Last year the author helped conduct a series of meetings for the EPA dealing with wetland restoration and creation in addition to the Association's 1986 national wetland symposium: Mitigation of Impacts and Losses (Kusler, 1988). There was consensus among the experts that hydrology was the most critical factor in restoration/creation. Without adequate hydrology, a project is bound to fail despite all the attention to soils, replanting, and subsequent management. And yet, hydrology is only superficially considered in many restoration projects with little effort even to consider the impact of long term hydrologic events on the wetland systems.

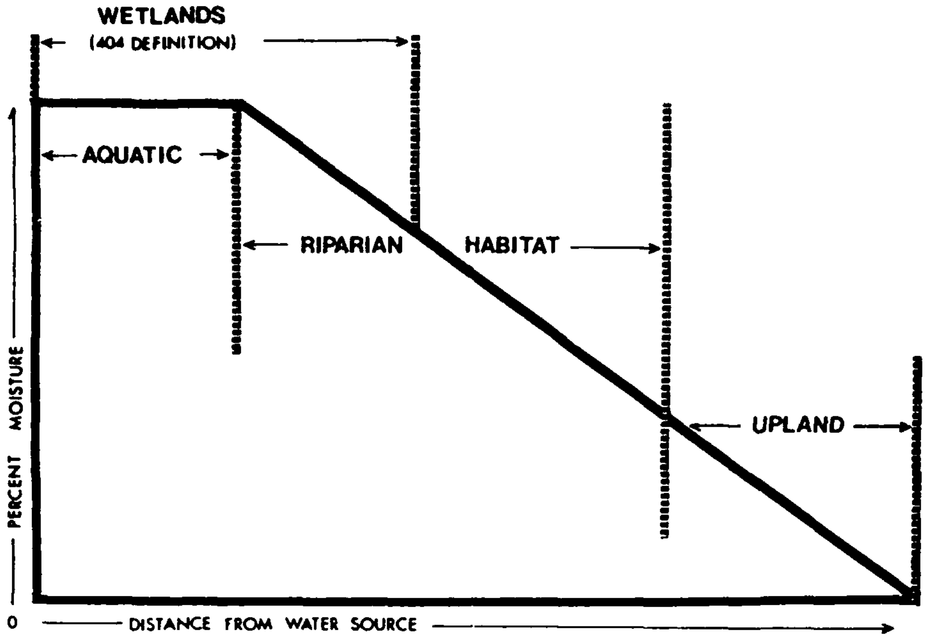
In the past, wetland managers have, in general, had little expertise in wetland hydrology. But, wetland protection and management has also been quite ineffective in many contexts. For example, the Office of Technology Assessment in 1984 (OTA, 1984) estimated that 90% of wetland losses are not controlled. This is, in large measure, because alterations (both individual and cumulative) in wetland hydrology and water quality from drainage, impoundment of waters, water extractions, and a broad range of point and nonpoint sources of pollution are subject to little or no regulatory control. Wetland protection and management efforts have been particularly ineffective in agricultural areas and in urban contexts where drainage and stream flows are highly modified (engineered) and watershed changes greatly affect the quantity and quality of runoff.

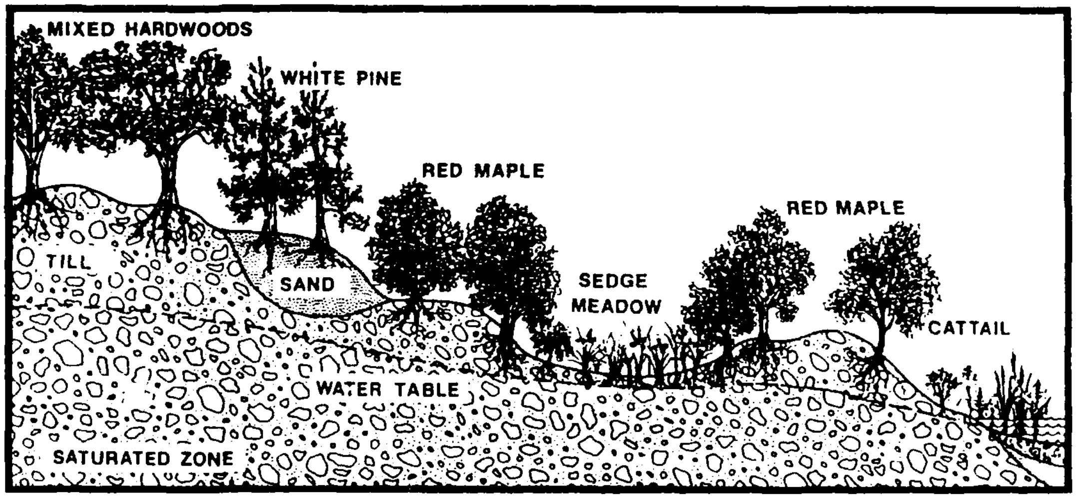
Wetland hydrology is relevant to many aspects of wetland management. Some include:

--Identification of an area as a wetland. The presence or absence of ground and surface waters and the frequency of inundation determine directly or indirectly the status of an area as a wetland or nonwetland (depending upon the wetland definition used) (See, for example, Duever et. al., this proceedings). Vegetation and soils reflect hydrologic conditions. In many instances a wetland may be identified through vegetation or a combination of vegetation and soils which imply a particular hydrology. Knowledge of hydrology, including flood events, becomes increasingly critical where vegetation or soils are ambiguous (e.g., certain riparian habitats, altered systems).

--Boundary determination. Evidence of surface and ground water elevations, frequency of flooding, and other factors is also needed for boundary determination, particularly where vegetation and soil conditions are ambiguous, a wetland has been newly created (no soil profile), or alterations have occurred in natural hydrology.

WETLANDS AND SOIL MOISTURE





SOIL MOISTURE AND PLANT COMMUNITIES

Such evidence may include direct measurements of ground and surface water levels, flood maps, stream gauging records, water marks on trees.

--Assessment of wetland functions. Hydrologic information is essential to assessment of certain wetland functions such as flood conveyance and flood storage, groundwater recharge and discharge, pollution control and sediment trapping, stormwater retention, and fisheries and waterfowl production. The expected longevity of a particular function is also dependent upon hydrology and ancillary factors such as rates of sedimentation. For example, wetland flood storage may be quickly destroyed by high rates of sedimentation.

--Assessment of the natural hazard potential of a site for a particular activity. The suitability of a wetland site for a particular activity depends not only upon wetland values but threats to the activity from natural hazards. Natural hazards depend largely upon hydrologic characteristics, including flood flows (frequencies, depths, velocities, ice in water), erosion potential, structural bearing capacity, subsidence potential, and liquefaction potential (these last two being dependent, in part, upon soil water).

--Determination of legal status of wetlands. The legal status of a wetland is often dependent upon its hydrologic characteristics. (See Dawson, this proceedings). Public or private ownership of the bed often depends upon whether the wetland is subject to the ebb and flow of the tide (public ownership) or is inland navigable water (public ownership). River beds and beds of smaller lakes are often privately owned, but like other water bodies and wetlands may be subject to "public trust". Courts have often held that private uses cannot infringe upon certain public uses for waters held in trust. The hydrologic values of a wetland are also important in determining whether regulations "take" private property. Regulations designed to prevent private landowners from increasing flood damages on other land, reducing water quality, or otherwise damaging adjacent landowners or the public are not considered a taking.

--Determination of short term and long term sensitivity to impacts. The sensitivity of wetlands to various impacts including recovery potential from such impacts is often closely related to hydrologic characteristics. For example, salt marshes and marshes along rivers with continuous water supply and periodic flushing by flood flows are quite "self healing", providing the wetland topographic contours are maintained. In contrast, pothole wetlands and other closed systems with perched water may be very sensitive to impacts and may have a high impact potential and a very slow recovery rate.

--Determination of the "persistence" of a wetland for planning or management purposes. The ability of a wetland to continue to function as a wetland over a long period of time is determined by long and short term sedimentation rates, periodic flushing and hydroperiod, and other hydrologic factors (See Niering, this proceedings).

--Design and evaluation of restoration/creation projects. Determination of wetland restoration/creation potential of a site, the specifics of project design, and maintenance and monitoring needs depend upon the hydrology of the site (See Vance, and other papers in restoration/creation section of this proceedings).

2. IS ENOUGH KNOWN (IN BASIC SCIENCE) ABOUT WETLAND HYDROLOGY TO GUIDE MANAGEMENT DECISION-MAKING?

There are many unresolved basic science issues with regard to hydrology in general, and, more specifically, with regard to wetland hydrology. These gaps in basic knowledge have led some hydrologists to characterize the entire status of wetland hydrology as "inadequate". On the other hand, many of these same gaps also apply to a broad range of non wetland engineering and construction activities including efforts to map floodplains, predict stream meander, design drainage ditches, and construct dams. Engineers, planners and others have not refrained from undertaking these efforts or constructing structures despite these uncertainties. What is known is applied with various "safety factors" to address partial gaps in knowledge.

Nor should such gaps prevent wetland managers from applying what is known about wetland hydrology despite some deficiencies in basic knowledge. A great deal of basic hydrologic information and hydrologic data relevant to assessment of wetland functions and the understanding and management of wetland systems exists for rivers, lakes, estuaries, the coasts and, more generally, for precipitation, runoff, and detention times. Little of this knowledge is being applied to wetland management. When it is applied, emphasis is often placed upon narrow water budget calculations and depth of water without consideration of other factors. The role of long term fluctuations in precipitation and water level and interactions of wetlands with hydrology are rarely considered.

Unfortunately, there is no convenient source book for the wetland manager interested in wetland hydrology. The "wetland" literature dealing with hydrology is meager, as pointed out by Clark et. al (1978) and Mitch and Gosselink (1986). Many of the studies are empirical and have addressed lakes and bogs and have focused on particular aspects of hydrology--primarily ground

water flows. Most of the available literature is topically specific and few articles deal with practical assessment needs. The wetland manager wishing to address wetland hydrologic issues must, to a very considerable extent, depend upon the broader literature dealing with hydrology and geomorphology.

Although wetland hydrologic references are relatively few in number, wetlands are part of the landscape and can be interpreted consistent with broader geomorphological and hydrologic principles. Some special problems are encountered with wetlands such as estimating the impact of vegetation on flows, determining whether or not a wetland is hydrologically connected to the groundwater, and estimating transpiration versus evaporation rates. But, otherwise most wetlands can be approached from a hydrologic perspective like other depressional features where surface or ground waters collect or flow.

3. WHAT ARE THE PRINCIPAL ISSUES OR TOPICS OF INQUIRY IN UNDERSTANDING THE RELATIONSHIP OF HYDROLOGY TO WETLAND CHARACTERISTICS AND FUNCTIONS?

Five principal questions arise with regard to the flow of water into, through, and out of a wetland:

- What are the major and minor sources of surface and subsurface water for a particular wetland?
- What are the magnitude and characteristics of these sources (depth, velocity, turbidity, dissolved and suspended materials, temperature, etc.) over a specified period of time including maxima and minima and mean (average) conditions?
- What plant, fauna, soil and other associations result or are associated with the sources and their specific characteristics?
- What happens to the water while in the wetland (e.g., reduced velocity and sediment deposition, nutrient removal, groundwater recharge, etc.)?
- What changes occur in the wetland and *to the wetland* due to the flow of water and the substances it brings in and out of the wetland?

4. WHAT ARE THE CRITICAL HYDROLOGIC FEATURES OF A WETLAND?

The types of hydrologic information needed for various management decisions of course differ depending upon a range of factors (as discussed below). Critical hydrologic features of a wetland may be broadly grouped into two categories: those relating directly to the water in the wetland and those which interact with the water to produce or affect certain characteristics or functions.

The Water

--Sources of water to the wetland; path of discharge of water from wetland. The source of wetland water or combination of sources (direct precipitation, ground water, surface water) is important from several perspectives. As discussed below, understanding of the source can suggest wetland origin. Source often indirectly determines specific wetland characteristics. For example, water entering wetlands from the ocean has, in general, some salinity and a high energy profile due to tidal action. This, of course, determines vegetation and fauna. In contrast, ground water is (relatively speaking) fresh, cold, free of sediment, and has little erosive force as it enters the wetland. River water often contains sediment and may have considerable velocity during floods.

Source of water may affect the actual ownership of the wetland since most wetlands subject to the ebb and flow of the tide are in public ownership unless title has been passed from the public to a private owner. Wetlands which are part of or connected to coastal or inland "navigable waters" are usually also subject to public trust.

The manner in which water is discharged from the wetland is also important. Discharge primarily through evaporation or evapotranspiration means that the wetland is probably an effective sink for sediment and very sensitive to nutrients, sediment, or dissolved or suspended substances. It will have limited flood storage or flood conveyance potential. In contrast, discharge through a surface outlet and active flow is less likely to be a sink and has increased flood conveyance potential. Discharge into the groundwater will increase the potential value of the wetland as a recharge area.

--Depth of water in various areas of a wetland. The depth of water in various portions of a wetland during "normal" periods, during long term fluctuations in precipitation, and during low frequency but extreme events such as floods is critical to many wetland characteristics and functions.

From a definitional and legal point of view, depth of water determines the "waterward" edge of a wetland vs. deepwater habitat in a lake, stream, or the ocean. For example, the U.S. Fish and Wildlife Service considers inundated lands to be wetlands to a depth of 6 feet. Depth of water may also determine land ownership and public rights in waters since the beds of most navigable

waters are either owned outright by the public or subject to public trust or navigable servitude.

In general, depth determines vegetation type and vegetated vs. open water areas of a wetland. Many plant species (particularly forested wetland species) are very sensitive to depth. Depth is also critical to use of particular portions of wetlands by muskrats, fish, wading birds, turtles, and other species. It is relevant to flood conveyance and flood storage potential. Recreational potential for some uses such as canoeing also depends upon water depth.

Water depth, of course, reflects the topographic contours of the wetland and the amounts and levels of incoming and outflowing waters including direct precipitation and surface and subsurface flows. Depth varies with precipitation and tide levels and also varies over time as deposition and erosion occur within a wetland.

Average depth at a particular point over a period of months or years may be most critical for certain plant and animal use. However depth during long term fluctuation and flood flows is critical from some perspectives (See Weller, this proceedings).

Depth under "average" conditions may often be inferred from vegetation type. Depth can be directly measured in a field survey through the use of a stadia rod or similar device. However, it is often difficult to decide what is "bottom" when the substrate includes several or many feet of unconsolidated organics.

Depth during long term precipitation cycles and flood events is harder to predict. Flood maps, stream flow records, and various modeling approaches based upon precipitation may be used.

--*Velocity of the water.* The velocity of the water entering and passing through a wetland determines the flood conveyance capacity of the wetland, the flood storage capability, the sediment transport capability of the water entering the wetland, the sediment trapping ability of the wetland, the potential for short and long term flushing of sediments out of the wetland, the pollution control capability of the wetland, and other features. In general, the higher the velocity of water passing through the wetland, the greater the flood conveyance capability. The higher the velocity of the water entering the wetland and the lower the velocity of the water exiting, the greater the sediment trapping and pollution control functions.

Velocity is also important to the short term and long term flushing potential of a wetland and to vegetative successional sequences (as discussed below).

"Guesstimates" of velocity in a wetland can be made based upon the overall characteristics of the wetlands and the adjacent water body. In general, very low velocities at all water levels can be expected in lakeshore and isolated wetlands and wetlands along larger, very low gradient streams (except areas impacted by storm waves). In contrast, relatively high velocities may be expected for coastal wetlands impacted by hurricane waves and for riverine wetlands along high gradient rivers and streams.

"Guesstimates" are also sometimes possible based upon examination of soils including the organic content and size of materials. Deep organic soils imply long term, low velocity conditions. Mineral soils, particularly those containing small rocks imply higher velocity.

The velocity of the water in a wetland can be more specifically measured directly or modeled during various levels of flow through a variety of techniques.

--*Wave action.* Wave action within a wetland is only an issue where the wetland contains substantial open water or the wetland is adjacent to open water with at least moderate depth (over four feet). Wave height and the erosive force of waves is dependent upon the depth of the water, the "fetch" (width) of open water, the presence or absence of particular types of vegetation in the water, and the substrate material.

Wave action determines, in part, the type and condition of wetland vegetation and soils. Wave action is often a major factor in bank erosion and may play a major role in flushing sediment, nutrients, and other materials from some wetlands. Most wetland plants cannot germinate and grow in moderate to high energy zones, particularly where this energy continues for long periods of time. However, some plants may survive in a least moderate energy areas if protected until maturity.

"Guesstimates" concerning wave energies may be made based upon air photos or topographic maps indicating open water areas and the depths of such areas. The lack of vegetation or wetland soil along the shores of lakes, larger rivers, or the ocean may also infer moderate to high wave energies.

Some coastal flood maps also identify areas at high risk from waves. Various models and other predictive approaches can be used to calculate potential wave energies at particular points.

--*Areas subject to ice.* Freezing soil temperatures and ice in shallow wetland areas located in the northern tier of states and at higher altitudes in mountains limit the growth of certain plants and help rework the nearshore soil profile. Ice may also affect habitat values, flood storage and conveyance capacity, pollution and sediment

control capacity, groundwater recharge and other functions for at least portions of each year.

Wetland areas subject to freezing and ice may be identified by field surveys during the winter and (guesstimated) through air photos and topographic maps indicating water depths.

--*Fluctuations in water sources including hydroperiod (duration) of various depths, velocities, sediment loadings, etc.* As will be discussed in greater depth below, fluctuations in the characteristics of the water flowing into and out of a wetland may be as important as "normal" or mean conditions. For example an area may or may not qualify as a wetland under various statutes and regulations (e.g., Rhode Island freshwater wetlands) depending upon the flood events and the frequency of such events during the year. The timing and duration of maxima and minima for water depths and velocities determine many wetland characteristics and functions. Relatively short term inundation by flood flows, which is common for coastal and riverine wetlands, may not have much effect on vegetation unless it changes salinities or causes erosion. In contrast, long term inundation of the sort common for isolated wetlands dependent in whole or in part upon the water table will often kill wetland trees and plant species.

Long term fluctuations in water levels and hydroperiod are often difficult to predict. Direct gauging of water levels is, of course, desirable. But long term records are rarely available. Flood maps, flood records, tidal records, stream and lake gauging records, records pertaining to ground water levels, long term precipitation records, and predictions concerning various magnitudes/frequencies of precipitation from NOAA can be used to a greater or lesser extent, depending upon the circumstances. Other types of site-specific inferential evidence can also be used such as water marks on trees, soil conditions, and condition and type of vegetation.

--*Dissolved and suspended materials in water (nutrients, sediment, detritus), turbidity, temperature.* Although dissolved and suspended substances in waters flowing into, through, and out of a wetland may not be considered a "hydrologic characteristic" per se, such materials play major roles in determining wetland habitat, food chain support, pollution control, and other functions. Suspended solids also affect flow rates and erosive force of waters. Waters with a very high sediment load flow more slowly through wetlands than clear water and also have less erosive force.

In a more general sense, suspended and dissolved substances determine the long term shape, size, depth and even location of the wetland and its long term functions. For example, high sediment loadings may fill a wetland, reducing or destroying its flood conveyance, flood storage, and virtually all other functions. Similarly, high loading of nutrients and organics may lead to rapid filling of the wetland by organic matter, altering its hydrology.

Water temperature and turbidity influence vegetation and fauna, affecting habitat values and indirectly affecting flood conveyance, pollution control, and other functions.

Features of the Wetland Which Interact With the Flow of Water

A wetland develops certain characteristics in response to its hydrology, which, in turn, affect its hydrology:

--*Size, shape, depth of the wetland.* As discussed further detail below, the size, shape, contours, and depth of most naturally occurring wetlands have been formed, in large measure, by the forces of ice or flowing water. The shape of the wetland determines, to a greater or lesser extent, its hydrologic and all other functions. For example, flood conveyance is highly dependent upon the shape of the wetland.

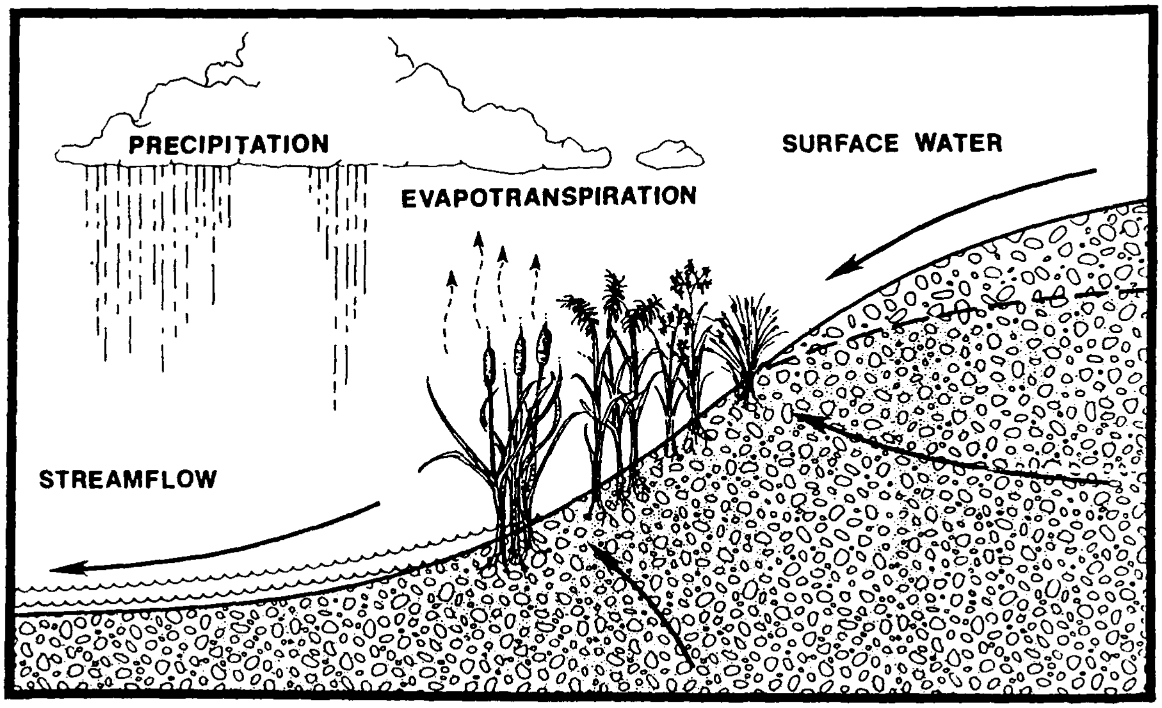
However, the size, shape, contours, and depth of a wetland are, by no means constant. The same erosional and depositional forces that created the wetland continue to work to modify its shape over time. Some modifications may be quite gradual, such as reduction in depth due to slow sedimentation. Others may be very rapid such as major erosion and flushing of a riverine wetland by a 100-year flood event. These changes may be accelerated by impacts to the watershed such as tree-cutting, drainage, and urbanization, which increase peak flood flows and sediment loadings.

--*Wetland vegetation.* Wetland vegetation is, in large measure, dependent upon hydrology. However, the vegetation also modifies the hydrologic characteristics of the wetland. Vegetation produces detritus and organic wetland soils which may gradually decrease wetland depths. The type and density of vegetation affects water velocities in the wetland and flood storage and conveyance potential. It determines, in part, sediment and detritus trapping potential. It affects erosion rates within the wetland. It may also affect ground water levels in the wetland through evapotranspiration (forested wetlands).

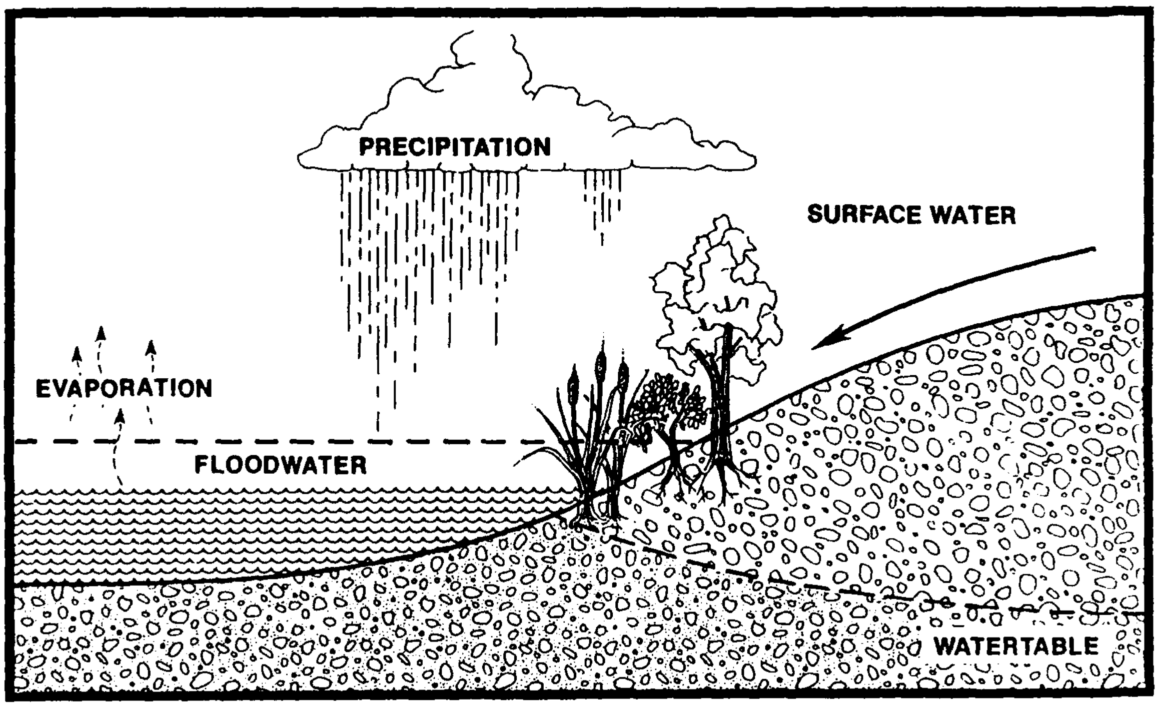
--*Wetland soils.* The type, depth, and other characteristics of wetland soils are determined, in part, by the suspended and dissolved substances in the inflowing waters. But soil type determines, in part, vegetation, which, in turn, affects the flow of water. Soil characteristics also determine, in part, the extent to which a wetland will remove or alter suspended or dissolved substances from the water. Soil characteristics determine potential erosion rates within a wetland during periods of

SOURCES OF WATER: FRESHWATER WETLANDS

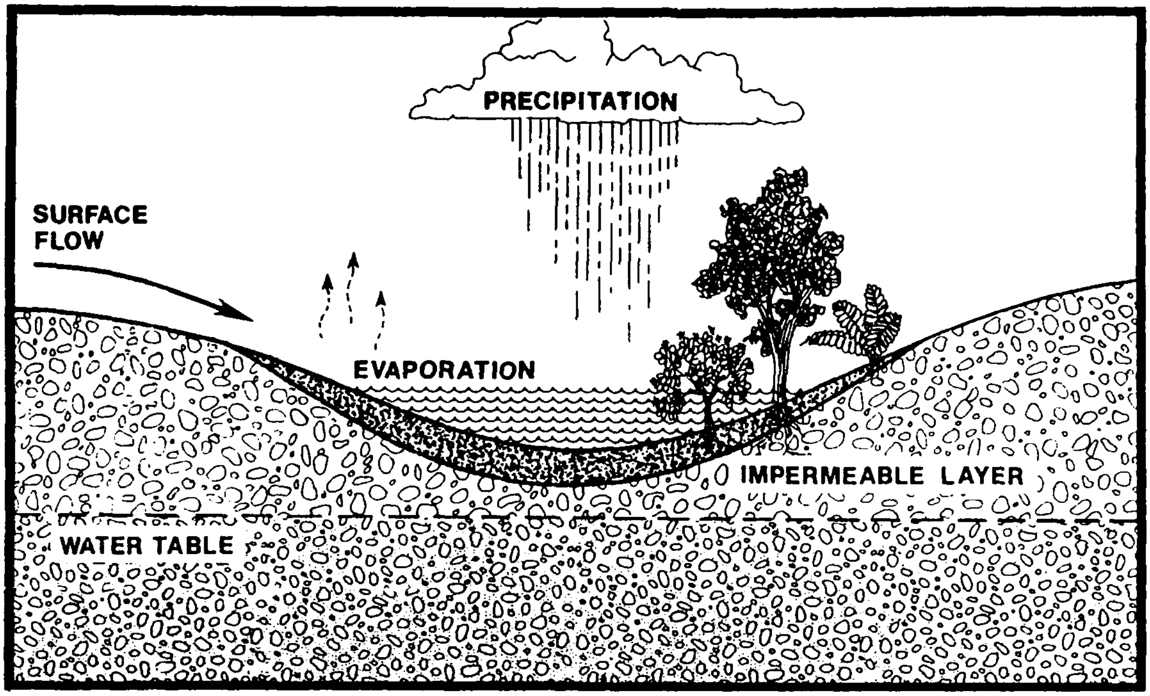
(Adapted from Novitzki, 1979 – figures by Richard B. Newton)



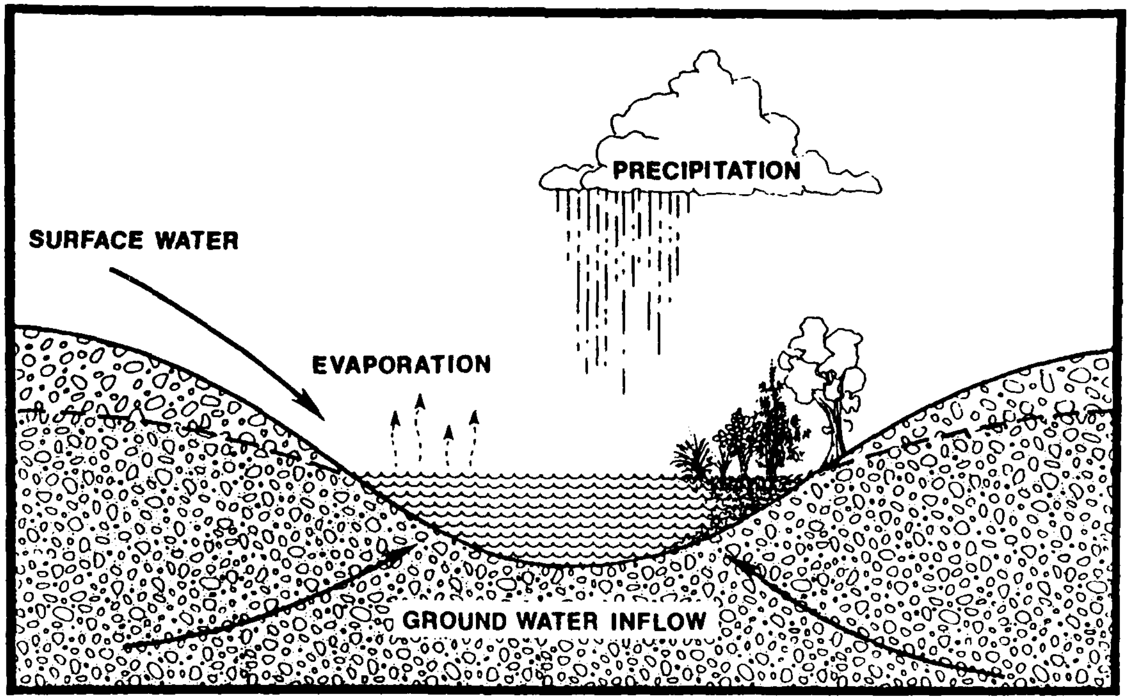
GROUND WATER SLOPE WETLANDS



SURFACE WATER SLOPE WETLANDS



SURFACE WATER DEPRESSION WETLANDS



GROUND WATER DEPRESSION WETLANDS

high flows.

--*Wetland fauna.* Animal life is, to a greater or lesser extent, interactive with wetland hydrology. Microorganisms and small organisms feed and break down detritus, thereby influencing wetland soils and wetland depth. Beavers, of course, substantially alter drainage. Muskrats and alligators deepen some wetland areas or reduce vegetation in others. Geese and other waterfowl may also essentially denude some areas of a wetland. Carp destroy aquatic vegetation and increase water turbidity.

5. CAN WETLAND HYDROLOGY AT A SITE BE EVALUATED THROUGH A SINGLE SET OF MEASUREMENTS OR A SINGLE SET OF CALCULATIONS RELATED TO ONE POINT IN TIME?

For some management purposes, a single, site-specific assessment of water depth, velocity, turbidity and other hydrologic characteristics may suffice to indicate immediate habitat, flood storage, or other potential. But a single set of measures is often of limited value in assessing the parameters listed above such as hydroperiod and the longer term functions, sensitivity to impacts, and restoration or enhancement potential.

Understandably, wetland managers and hydrologists hired to assist them often focus only upon intermediate or small scale hydrologic events and assume a steady condition for the wetland shape, size and substrate. This is the least expensive and simplest solution and suffices for some purposes. But, as pointed out below, such a focus may provide a very incomplete view of wetland functions over time. Spending large sums of money on detailed water budgets for small scale and intermediate events may also be a waste of money if the critical events determining long term wetland functions include hydrologic extremes (as is often the case).

Consider, for example, an effort to determine a wetland's ability to trap sediment or certain pollutants. A detailed short term water budget may be prepared with emphasis upon inflow and outflow and detention times. This information, along with some other assumptions, can be used to estimate the ability of a wetland to trap suspended materials. But managers are generally left with the conclusion: "Yes, a certain amount of sediment, toxic material, etc. will be trapped in the wetland. But substantial uncertainties exist as to what will happen to this material over time." A consultant hydrologist may assume total retention or, on the other hand, that sooner or later the materials will leave the wetland through sediment transport or resolution, providing no net long term pollution control.

But either is likely to be an assumption. Some sediment and detritus will ultimately leave the wetland, depending upon long term anticipated velocity in a wetland during flood stages, and other factors. Even in contexts where the sediment may be removed by erosion during large flood events, the old adage applied to more traditional types of pollution that "the solution to pollution is dilution" applies (with some qualifications) here as well. The damage to wetland vegetation, fauna, and other functions caused by various types of pollution depends not only upon the ultimate fate of substances but also, when they leave the wetland, their form, dilution factors at time of exit, and the sensitivity of target fauna at the time of release. To estimate these factors, hydrologic analyses including estimates of erosion and sediment transport potential are needed for various flood conditions since most flushing of sediments from a wetland may take place during these conditions.

Calculations pertaining to "extreme events" and focus upon long term fluctuations in water levels and the role of extreme events can suggest long term erosion and sediment transport potential and the long term fate of temporarily (or permanently) trapped materials. They can also suggest whether the release of the materials might, at this time, have severe impact upon wetland functions and values. For example, release of sediment and trapped toxic materials from a wetland during low flow periods might have severe impact upon upon downstream flora and fauna. But release during extreme flood events when erosion is greatest (the most likely) may have little impact because of the extreme dilution factor and the time of the year when high flows typically occur (early spring, late fall).

6. HOW CAN A WETLAND MANAGER BEST MAKE USE OF "HYDROLOGIC" EXPERTS?

Unfortunately the average "hydrologist" may not *initially* be able to help you, as a manager, in evaluating a permit for a proposed activity, preparing a wetland management plan, or carrying out other calculations. This will, of course, depend upon the issues and the qualifications of the hydrologist.

The term "hydrologist" is a rather imprecise term. It is broadly used to describe individuals having expertise in calculating the flow of water under various conditions and the impact of such flow upon various structures, landscapes, etc. Hydrologists have many academic backgrounds and highly varied work expertise. They include geologists (geomorphologists, hydrogeologists), engineers (water resources, sanitary, etc.), geographers, and meteorologists.

The perspectives and actual expertise of these individuals also vary greatly. In my experience, the geologists are likely to have a long time dimension view of a problem and at least some

understanding of geomorphic processes and water/sediment relationships but have rarely had much training in site-specific water budget calculations, flood flow calculations, etc. The engineers are likely to have a much shorter time perspective on a problem and have considerable training in calculating site-specific runoff and project design (dams) but often know little if anything about water/sediment relationships or water/ geomorphological relationships. Most "hydrologists" are likely to have specialized in particular problems or issues since college and this may have narrowed their perspective but increased their expertise in one area only. For example it is common for a water resources engineer or hydrogeologist working with ground water to have limited experience with surface waters. Finally, most "hydrologists" have never had even a single course in wetland management nor are likely to have any idea of how particular hydrologic maxima, minima and "means" may relate to particular wetland vegetation, fauna, or functions.

This is not to criticize "hydrologists". The wetland manager, however, must be able to ask the right questions in hiring a consultant and must be able to work with the hydrologist to refine issues and determine the implications of various hydrologic conclusions. Fortunately, an increasing number of hydrologists (like those represented at this symposium) have already become aware of the wetland-specific management issues.

7. DOES THE ORIGIN OF A WETLAND PROVIDE CLUES AS TO ITS HYDROLOGY?

Knowledge concerning the origin of a wetland can provide some initial clues as to the dominant and secondary sources of water for a wetland, retention time for waters, retention time for sediments and other materials deposited in the wetland, and the short and long term potential for modification of the wetland shape, size and even location by hydrologic factors.

Wetland origin can be inferred by size and shape and general location. For example, most wetlands in river channels or on floodplains were created by riverine erosional/deposition forces. Most smaller nonriverine wetlands in the northern tier of glaciated states have been created by the melting of ice blocks in glacial till, outwash, or moraines. Most coastal wetlands behind barrier islands were created by coastal erosion/deposition forces. Estuarine wetlands are usually the result of a combination of coastal and riverine erosion and depositional processes.

With the exception of man-made wetlands, most wetlands are not only dependent upon water for their continued existence as wetland but *have been created* by the forces of water. The water/wetland relationship is not passive, as discussed above. Relatively infrequent but large scale hydrologic events determine, to a considerable extent, the location, size, shape, and even the *vegetation* type for the wetland. In turn, vegetation and biological processes in the wetland determine (in part) the flow rate through the wetland and what happens chemically to the water. Wetland vegetation, to some extent, slows the passage of water through the wetland. Beavers (part of the wetland biota) may slow the passage of water even more dramatically.

Naturally occurring wetlands have been created in three principal ways:

- Melting glaciers.
- Erosion and deposition by rivers.
- Coastal erosion and deposition processes combined with rising and falling sea levels.

A variety of other lesser natural processes are also important:

- Beaver.
- Solution (karst topography).

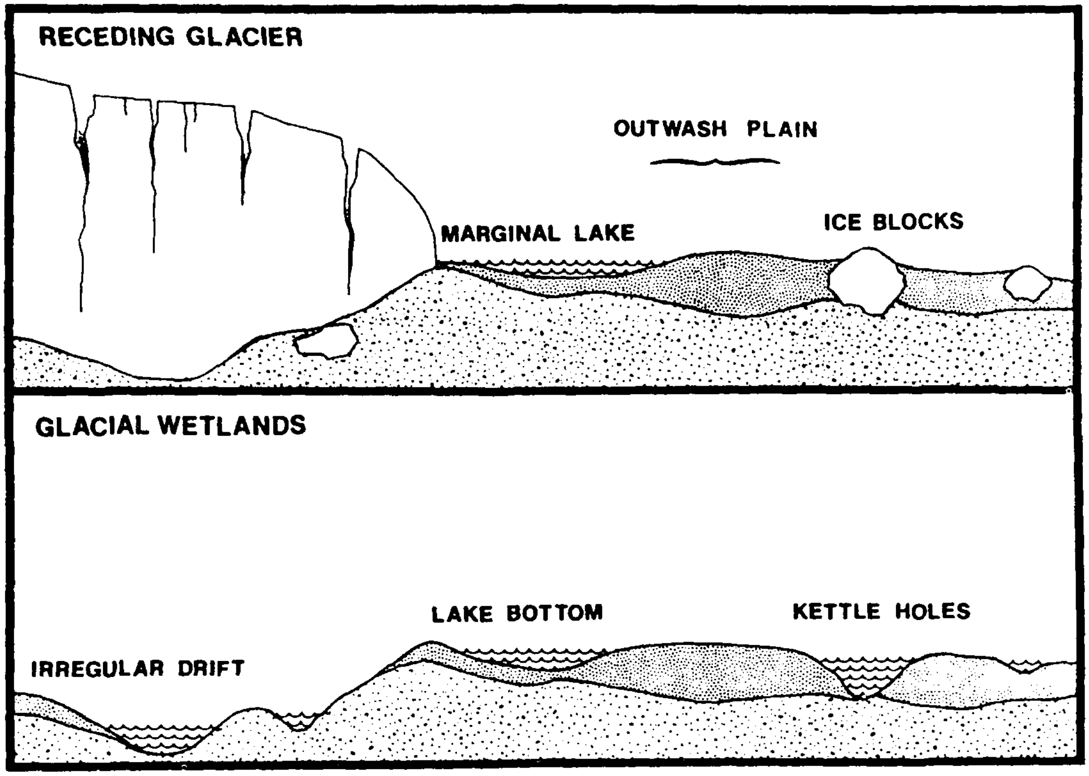
In addition, wetlands have been created or enlarged in size by a variety of human activities.

A brief discussion for each type and some of its hydrologic characteristics follows:

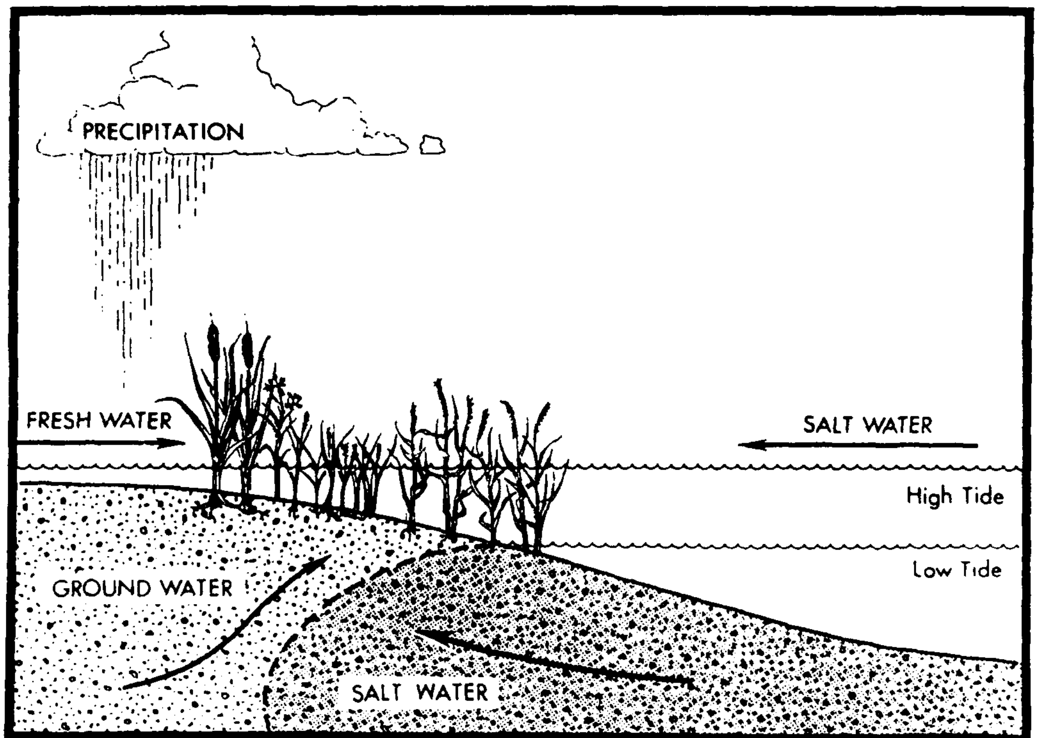
Glacially formed wetlands in till, outwash, lake deposits or bedrock. The majority of wetlands (literally hundreds of thousands of wetlands) in the northern tier of states from Maine to Washington state and at higher altitudes in mountains further south were created by retreating glaciers 9-12,000 years ago. Glaciers created wetlands in several ways.

First, glaciers left melting chunks of ice in glacial till and outwash. Upon melting, pits and depression were formed in the landscape. These filled with water and lakes and wetlands were formed wherever the depressions intersected the ground water table or where fine clay or organics sealed the bottoms. The majority of smaller wetlands in the northern U.S. including the wetlands of the "pothole region" were formed this way.

Many of these wetlands are partially or wholly fed by ground water. Direct precipitation may also be important. Surface runoff is a secondary source because of the small size of the watersheds. Some of these wetlands, but not all, have outlets. Many are subject to dramatic short term fluctuations in water levels due to seasonal variations in precipitation and runoff and long term fluctuations due to longer term precipitation cycles. Nearshore areas are also often subject to seasonal ice action.



GLACIERS CREATE WETLANDS



TIDAL WETLANDS

Closed basin wetlands and those with limited outlets are among the most sensitive wetlands to nutrients, sediment, toxics and other materials because they are sinks and are not flushed by periodic high velocity flows as are riverine and coastal wetlands.

A second type of glacially created wetland is that of a glacial lake. Glacial ice may either have blocked drainage directly or the glacier may have created a moraine which blocked drainage, forming a lake. Some glacial lakes were thousands of square miles in area (e.g., Lake Aggasiz). Once the ice retreated or the moraine dams eroded (downcutting outlets by drainage water), the lakes partially drained, creating extensive wetlands. These wetlands are often underlain by varved lake clays. Former glacial lakes form some of the largest forested wetlands in the northern regions and often contain deep peat deposits.

Remnants of former glacial lakes are primarily fed by surface runoff in small or larger channels and by direct precipitation. They, like their smaller brethren, are subject to short term fluctuations in water levels due to precipitation and to longer term fluctuations in ground water levels due to longer term precipitation cycles. Often a portion of the wetland remains open water. They are typically sinks for sediment, toxics, and nutrients and are not subject to high velocity, erosive flows. They appear to be less sensitive than small pothole wetlands to water quality changes due, in part, to their large size.

Glaciers created wetlands in a third way when they scooped out and scoured river valleys and bedrock deposits creating, in some instances, large and deep lakes with wetlands along their margins such as the Great Lakes and the Finger lakes. They also created shallow depressions in bedrock including some prairie potholes and many wetlands on the Canadian shield.

Wetlands created by scour are often very low in nutrients (due to limited soil in the wetland and watershed). They are more directly fed by surface runoff than their small, "pothole" counterparts. But watershed areas are also generally small.

Wetlands along the Great Lakes and other major lakes are subject to periodic changes in water level, depending upon basin precipitation. Fringing wetlands often migrate shoreward or lakeward, reflecting these levels. They are periodically subject to storm waves and ice action and may be flushed by these processes. They are, therefore, less sinks than their smaller, groundwater-dependent counterparts. But, they are dependent upon water quality changes in the lakes as well as direct sources from the adjacent lands.

Wetlands on wave protected coastal lowlands. A second principal group of wetlands was formed by tidal and coastal storm processes (often combined with land-based stream erosion and deposition). In general, coastal and estuarine wetlands occur wherever a gently sloping beach of sand, gravel, silt, or other particulate matter is protected from wave action by a harbour, barrier island, reef, or gently sloping shore (e.g., Louisiana). The principal source of water for these wetlands is the ocean. However, freshwater inflows from rivers and streams, surface runoff, and groundwater may play important roles in determining vegetation and fauna. Lower-lying wetlands are transected by tidal channels carrying tidal (often combined saline and fresh water) inflows and outflows twice a day. These wetlands are subject to storm wave and ice action (northern areas).

Under natural conditions, coastal and estuarine wetlands move seaward and inland as sea levels fluctuate. Deposition of sediment, organic matter, and peat in some wetlands permits some adjustment to water level changes.

Coastal and estuarine wetlands are less likely to be long term sinks for sediment, nutrients, organic material, and toxics than glacially formed freshwater wetlands because they are subject to daily tidal flushing. These wetlands also often undergo severe erosion and deposition during hurricanes and winter storms.

Wetlands in river channels and on river floodplains. The remaining major wetlands of the U.S. are located along and in the channels of rivers and on the floodplains of rivers. River channel and floodplains are formed by a combination of erosion and deposition during floods and more "normal" times. Major wetlands are found in the channels and on the floodplains of mature streams with low gradients. Many occur in "oxbows"--former river channels. Smaller, thin ribbons of wetlands are found along smaller, high gradient streams.

With the exception of backwater areas, riverine wetlands are periodically flushed and scoured by major floods. Most channel and near-stream riverine wetlands lie within floodway areas.

Many riverine wetlands are very active, particularly in headwater areas. Water/sediment relationships are critical, particularly along high velocity rivers and streams. Deposition of sediment often occurs in relatively small flood events--1-5 year frequency flows. Massive erosion and down-cutting often occurs with large-scale flood events (50-500 year events).

The major source of water for riverine channel wetlands is usually the river itself. However, runoff from tributaries and surface runoff from adjacent lands may be major contributors to floodplain wetlands. Groundwater

may also be very important and the dominant source of water for wetlands located on river slopes or terraces.

Water velocity and erosion/deposition rates play significant roles with regard to wetland shape, size, depth, vegetation, and a broad range of wetland functions. Wave and ice action may be important on major rivers and ice action in northern areas.

Riverine wetlands are, to some extent, sinks for nutrients, sediment, and other materials, much like their lake and tidal counterparts. But periodic flushing is likely during stream meander and major flood events, particularly for river reaches with relatively high gradients and high velocity flows.

Wetlands created by beaver dams. Beavers are playing an increasingly important role (as they once did) in creating wetlands in northern forested areas of the Nation. Beaver dams are located on smaller streams and drainageways, often at the sites of preexisting wetlands. By blocking flows, beavers often increase the wetland area. Beaver dams are relatively ephemeral features (a hundred years at most) but cycles of wetland and forest often occur at the sites of beaver dams.

Beaver dam "wetlands" are fed primarily by streams and surface runoff but ground water sources may be important in some instances. Flow velocities are relatively low. Ice action may be an important factor. Beaver-created wetlands are partial sinks but organic materials and sediment may be partially flushed by a combination of flood flows, erosion, and organic matter decomposition.

Wetlands created by miscellaneous natural processes. Wetlands are also formed by a variety of other natural processes with varying hydrologic characteristics. For example, wetlands in the sand hills of Nebraska have been formed, in part, by the action of wind. These intersect the groundwater table and are groundwater controlled. The Everglades reflect a unique flow of surface and ground water at and directly below the surface with great sensitivity to human activity. Many small wetlands are found in "sink holes" and other solution formations in Kentucky, Indiana, and several other states. In general, water levels are groundwater dependent.

Wetlands created by human activity. Man has created as well as destroyed many wetland areas although the hydrologic characteristics of man-made wetlands are often quite different from those of wetlands created by the forces of nature. In many instances (reservoirs, blocked drainageways), man-made wetlands occupy sites of previous, naturally occurring wetlands. The new wetlands may have stabilized water levels, however. Reestablished wetlands are also typically marshes, replacing many shrub and forested wetlands

Except for wetlands forming in pits and depressions created by mining, sand and gravel operations, and other excavations, most man-made wetlands have been created by intentional or unintentional blockage of diffused surface waters or streams. These include waterfowl impoundments and wetlands along the margins of reservoirs created by dams, and wetlands in a wide variety of locations created by blockages to drainage by roads, railroads, dikes, levees and other fills.

Unlike naturally occurring wetlands, many man-made wetlands (particularly stormwater facilities and those created by roads, highways, levees, other fills) are subject to sediment loadings due to high rates of erosion in their watersheds. Many are sinks since they are not flushed, like their riverine, lakeshore, coastal, and estuarine counterparts by periodic high velocity flood flows. Wetlands created by water control structures (dams, dikes) are also likely to trap large quantities of sediment which would ordinarily be flushed downstream during normal flows or flood events.

8. WHAT FACTORS DETERMINE THE AMOUNT OF HYDROLOGIC KNOWLEDGE NEEDED FOR A PARTICULAR MANAGEMENT DECISION?

As one would expect, the amount of hydrologic knowledge needed for a management decision varies greatly depending upon a variety of factors. A few of the most important include:

The nature of proposed activity and its degree of impact upon wetland hydrology. The amount of hydrologic knowledge needed is greatest for wetland creation and for activities such as drainage, water impoundment, or water extractions which affect the quantity, velocity, hydroperiod, or other characteristics of water flowing into or out of a wetland.

Degree of alteration in wetland hydrology which has already occurred for a wetland or wetland watershed. The greater the alteration that has already taken place in a wetland or in its sources of water, the greater the need for an understanding of water sources to assess functions and impacts of further changes. Existing plant and animal communities may not reflect relatively recent hydrologic conditions, giving false indications of functions and values.

Type of wetland. An understanding of hydrology is important for all types of wetlands but is particularly critical for inland wetlands with "fragile" sources of water susceptible to diminution or destruction by even modest changes in the watershed or sources of water. These include wetlands fed primarily by diffused

surface runoff (no channel), those adjacent to small streams, and those fed by widely fluctuating ground water (e.g., some "potholes").

Management goals. Many wetland management statutes require that the wetland manager assess the impact of hydrology upon a proposed activity and not simply the impact of activity upon hydrology. For example, many wetland regulations state as one goal "the reduction in losses from flood or erosion hazards". The suitability of a wetland site for a particular activity is dependent not only upon the functions and values of the wetland but the flood, erosion, liquefaction, and earthquake hazards which may threaten the proposed activity or be aggravated by the activity. These hazards are due, in part or principally, to "hydrologic" and "hydraulic" features of the site which have long been considered by floodplain managers but often ignored by wetland managers. Consider, for example, the suitability of a riverine wetland as a residential development site. If existing wetland values alone were considered and few such values existed, the site might be considered suitable for development. But if the natural hazards were considered, the site might be outright dangerous for such development without extensive and expensive protective measures.

A variety of "hazard" related models and sources of hazard information are available to the wetland manager to help evaluate potential hazards and establish standards for activities in wetlands. For example, over 17,000 communities have adopted floodplain zoning, subdivision controls or other regulations. These regulations include both flood maps and standards for activities.

9. IS UNQUANTIFIED HYDROLOGIC DATA OF ANY VALUE TO THE WETLAND MANAGER?

In an ideal world, site-specific and quantitative hydrologic data would be developed for every wetland. This is unrealistic, however, because of the high costs and manpower and the long time frame required for evaluation. There as also limits to what can be achieved by short term quantified measurements. For example, it may take, at a minimum, several years of monitoring with a series of wells to determine ground water relationships.

For the purposes of legislation, planning, regulation, acquisition, and other management activities, wetland hydrology can be approached with varying levels of generality and quantification:

--General, unquantified presumptions based upon wetland origin and type, location in the watershed, or other factors. Generalized scientific information can usefully be used for some purposes, although generalizations must be applied with care. For example, it can be presumed that virtually all coastal wetlands subject to the ebb and flow of the tide and all riverine wetlards at or near the elevation of major rivers and streams are periodically flooded to considerable depths and, in some instances, with considerable velocity. This presumption is well-supported by a comparison of wetland maps and FEMA flood maps which indicate typical 100 year flood elevations of 7-14 feet above sea level for coasts and 5-15 feet above "normal" water levels for rivers.

Similarly, it may be presumed that wetlands are flood storage areas in many contexts.

Such general presumptions concerning wetland hydrology may be useful in enacting overall plans and statutes and defining "critical issues" needing more detailed data gathering or study. Such presumptions may also be used to shift the burden of data-gathering to those proposing to alter or create a wetland. However, general presumptions are not very useful in more specific permitting or in restoration/creation which require much more detailed information.

--More specific unquantified evaluation of the functions of a particular wetland based upon its location, its shape and size, the topography of the surrounding land, and other "site specific" and readily observable factors. The WET system and other relatively broad-brush assessment systems permit a generalized evaluation of wetland functions (in a relative, nonquantitative sense) based upon wetland-specific factors. Although quantitative values are not provided, this analysis establishes more specific presumptions for planning, acquisition, or certain regulatory purposes and for defining topics needing more specific study in regulation. Unquantified assessment is, in general, of limited value in defining specific mitigation needs (restoration or creation) or defining specific quantified permit conditions for permissible impact upon flood conveyance, etc.

--Site-specific (and, in some instances), quantitative evaluations through modeling or other approaches. Quantitative evaluations require calculations based upon the following sorts of data (1) existing data (e.g., flood and stream flow data, topographic maps), (2) air photo interpretation (3) superficial, one time field surveys (e.g., measurements of wetland depth), and (4) hydrologic (time series) monitoring through water depth monitoring, stream flow gauging, ground water monitoring, etc.

Site-specific, quantitative evaluations are needed for certain types of permit conditions (e.g., compensatory flood storage requirements), and for restoration/creation. There are limits even here, however, to the quantified calculations that can practically be carried out.

10. IS STABILIZATION OF WETLAND WATER LEVELS DESIRABLE?

A common objective of water project planning is to stabilize highs and lows in water levels (flood flows, low flows). Water levels and other hydrologic characteristics of rivers and lakes are often unintentionally or intentionally stabilized through dams, reservoirs, dikes, and channels.

Stabilization of water levels may also enhance certain wetland functions such as waterfowl nesting for at least a period of time. But, stabilization at low water levels, like that caused by channelization, may destroy wetlands outright. Stabilization by dams and outlet structures on lakes may also lead to the ultimate destruction of wetlands or classic "successional" sequences in vegetation rather than cycles. Continued extreme events are important to wetlands for several reasons:

First, some wetland functions such as fish spawning may depend upon period high water conditions. Without high water, certain wetland plant species may also not regenerate. For example, the western cottonwoods are being lost along streams with many dams and little flooding because cottonwoods depend upon flood flows for germination of seeds.

On the other hand, some functions depend upon period low water conditions. For example, a wetland may not store much floodwater if it is already filled with water.

Second, the importance of infrequent but severe hydrologic extremes to the "persistence" (long term maintenance) of wetlands is now widely recognized (e.g., Niering, this proceedings). For example, hurricanes and winter storms play primary roles in shaping floodplains, barrier islands, and beaches and the wetlands that lie within or behind such features. These infrequent but powerful flood events (in some instances accompanied by ice) not only sweep and scour sediment from wetlands but rip out trees and shrubs, returning wetlands (in some instances) to a denuded or semi denuded condition. At a minimum, many of the larger trees and shrubs may be killed or severely damaged. The classic vegetative progression of marsh to shrub swamp to forest may begin again and progress until the next major storm. Without such periodic flood or other catastrophic events, classic vegetative sequences (succession) would take place and wetlands would gradually disappear.

Long term (30 year, 90 year, 1000 year) cyclic fluctuations in precipitation which are reflected in major changes in groundwater levels and lake levels (e.g., the Great Lakes) may be equally important to the long term survival of smaller wetlands including the millions created by glaciers (discussed above). Many of these wetlands periodically become dry or water levels become very low, killing wetland shrubs and trees and allowing compaction and decomposition of peat and organic materials. Fires may also burn the peat. When ground water levels rise again, an open water/marsh wetland may result, beginning the cycle again.

11. FROM A HYDROLOGIC POINT OF VIEW, ARE BUFFERS NEEDED FOR WETLANDS?

Apart from habitat value and water quality protection, the most compelling reason for establishing a wetland buffer (e.g., 50 feet, 100 feet, 200) is based upon hydrologic functions and long term hydrologic characteristics of wetlands.

As noted above, the water level in wetlands varies seasonally and over longer periods of time with long term precipitation cycles. A buffer permits a wetland to migrate horizontally as water levels rise. Otherwise, the wetland may "drown" at high water levels and flood damage may occur to structures and other activities in the flooded area.

The need for buffers is particularly acute on the Great Lakes and for many groundwater-fed smaller lakes where long term fluctuations in water levels may exceed three to four feet. It is also acute for estuarine and coastal wetlands subject to subsidence or sea level rise. As several recent studies by the EPA indicated, most coastal and estuarine wetlands will be severely damaged or destroyed by the several feet of sea level rise projected by many wetland scientists unless the wetlands are able to migrate inland onto higher ground.

But long term fluctuations in water levels are not the only issue. The flood storage and conveyance functions and values of a wetland are absolutely dependent, in most situations, upon protection of the bank area outside of the normal wetland boundaries. As discussed below, the bank area acts like the walls of a reservoir during floods. Without it, the wetland has little or no storage potential in case of a major flood event.

12. IN WHAT CIRCUMSTANCES IS SEDIMENT A THREAT TO WETLANDS?

Apart from problems with water quality and fills, perhaps the number one threat to naturally occurring wetlands in watersheds heavily impacted by man is sediment. The build-up of mineral sediments is a great long term threat to wetlands and much more serious than the naturally occurring build-up of detritus, peats, and other organic material. Organic material can be broken down over time by bacterial activity, by periodic dessication and oxidation during low water or drought periods, and by fires. Mineral sediments will not, in general, be broken down (or very slowly) and must be mechanically removed by erosion or organisms if they are ever to leave

wetland systems.

In general, the sedimentation rates (in unaltered watershed) for naturally occurring "pothole" and other isolated depressional wetlands are low. Otherwise, these wetlands, which are usually 9,000 to 12,000 years old, would have filled long ago. But shoreline and watershed activities of man change this situation.

Sediment from shoreland or watershed sources is also a threat to most man-made wetlands including gravel pits, waterfowl impoundments, stormwater facilities, and those created by blockages to drainage (roads, fills, dikes). These wetlands are not in erosion and depositional equilibrium, as are many naturally occurring riverine and coastal wetlands. They also generally lack major flushing mechanisms. Many have been created in watershed areas with high levels of development and limited vegetation.

Although excessive sedimentation is a threat to many wetlands, moderate sediment levels in inflowing waters may be beneficial and even critical in some estuarine and riverine contexts. Land loss in coastal Louisiana has been attributed to lack of replenishing sediment from the Mississippi River due to channelization and the extensive upstream reservoir system (Templet, 1986). Interception of sediment by reservoirs may also destroy many other estuarine wetlands which might otherwise accrete at a rate consistent with sea level rise (as long as such rises are moderate).

Moderate sediment loadings in river waters are also important to many wetlands and riparian habitats located on the floodplains of high gradient, high velocity western streams. Under natural conditions, erosion and deposition on the floodplain of such streams is in approximate equilibrium. This equilibrium is disturbed if upstream reservoirs remove much or most of the naturally occurring sediment. The erosive force of water is increased as the amount of sediment is decreased (as anomalous as this may seem). The deposition is also decreased. The result is rapid erosion with a stream cutting down through the floodplain. Groundwater tables on the floodplain are lowered, destroying wetlands and riparian habitat. Stream bank erosion may also then increase.

13. CAN THE GROUNDWATER DISCHARGE/RECHARGE POTENTIAL OF WETLANDS BE QUANTITATIVELY CALCULATED?

Various hydrologic and geohydrologic field survey and analytical techniques can be used to calculate whether a wetland is discharging or recharging to the groundwater and the amount and location of this recharge and discharge. However, application of such techniques (installation and monitoring of piezometers, permeability and porosity studies for soils, water budget calculations, etc.) to even one small wetland can be extremely expensive and time-consuming. In addition, such analyses may be inconclusive with regard to longer term recharge/discharge potential at particular points along the wetland margin or beneath the water surface because of constant changes in wetland and ground water levels.

Early in this paper it was noted that some hydrologists have characterized the general status of knowledge concerning wetland hydrology as "inadequate". Such a characterization is particularly likely when hydrologists focus upon groundwater/wetland relationships. The general lack of research and understanding is due, in part, to the high costs of such research. The complexities of wetland systems reduce the transferability of groundwater/surface water research conducted in one wetland to others. Hundreds of books and reports and perhaps thousands of technical papers address broader groundwater issues. Some have relevance in highly specific contexts. For example, a paper dealing with groundwater movement of a toxic chemical in a particular soil or rock substrate would potentially be of some interest where groundwater contaminated with such a chemical were "up gradient" (in terms of the piezometric surface) from a specific wetland.

However, very limited research has been conducted pertaining to the rates and amounts of recharge and discharge into particular types of wetlands in specific contexts. Of the handful of studies available, most deal with peatlands or lakes--areas which are continually wet. Few deal with wetlands or riparian habitat in dry areas where recharge may be particularly important including ephemeral wetlands (wet only part of a year or only some years) and wetlands with highly fluctuating seasonal or long term water levels.

Net groundwater recharge or discharge for a specific wetland has most often been estimated (without any direct measurements) by completing the other "knowns" in a wetland water budget analysis and attributing the remaining water to groundwater flow. This is a relatively inexpensive approach and does not require time consuming and expensive field monitoring. But estimates are typically subject to large errors since the other calculations in a water budget analysis (direct precipitation, surface inflow and outflow, evapotranspiration) are also subject to large margins of error. In addition, water budget calculations provide estimates for net groundwater recharge or discharge, not actual recharge or discharge. Actual discharge and recharge may be much greater where both processes take place within one wetland.

Given this complexity and the lack of studies,

it is not surprising that a great deal of confusion exists concerning wetland groundwater recharge and discharge. Traditionally wetlands were often cited as recharge areas. It was argued that waters collected in these areas after rainfall and slowly seeped into the ground. More recently, wetlands have been broadly represented as discharge areas.

The broader hydrologic literature and studies to date suggest that most wetlands are discharge areas under natural conditions but that many may function as both discharge and recharge areas and a few may be predominantly recharge areas. Water may discharge from the ground along one side of a wetland (the high gradient side) and then reenter the ground at the other side (the low gradient side). Or a wetland may briefly serve as a recharge area when water levels are high (after a rain) and as a discharge area at other times. It seems likely that many ephemeral wetlands, which are dry much of the year, serve as temporary recharge areas after snowmelt or rains.

It also seems likely that some wetlands which are, under natural conditions, discharge areas may serve at least a part of the year (e.g., late summer) as recharge areas when the groundwater table is pulled down by pumping for municipal or industrial water supply or agriculture.

As with flood storage, *all* elements of the landscape except for highly impermeable surfaces may function, to some extent, as groundwater recharge or discharge areas or both. Typically groundwater discharge occurs when the groundwater table intersects a slope (hillside, streambank) or depression (river, lake, stream, wetland, gravel pit, etc.) with a surface elevation lower than that of the groundwater table and water is able to reach the open surface. Recharge occurs when precipitation falls on the land surface or a wetland, lake, stream or other water body at an elevation above the water table and the precipitation or water from the water body is able to move into the groundwater.

If the relationship between wetland water levels and adjacent groundwater levels were direct and steady and the permeability and porosity of the soil were known, studies of recharge and discharge relationships would be relatively simple. An analysis of ground water (piezometric surface) elevations versus surface water levels combined with a soil analysis would be enough to determine whether the wetland was recharging or discharging and how much it was recharging or discharging. But determination of piezometric surface is not easy and usually requires the drilling and monitoring of test wells. Determination of recharge and discharge relationships is further complicated by fluctuations in both wetland water levels and ground water levels throughout the year. Wetland water levels are typically high in the spring and fall and low in the summer. Groundwater elevations also vary with precipitation but with a much longer response time (e.g., weeks, months).

Calculation of recharge relationships are further complicated by the highly varied permeability and porosity of glacial tills, outwash, lake clays, organic layers, and other soils underlying wetlands. Another complicating factor is that semi-impermeable layers consisting of organic matter and fine clays develop along the bottoms or portions of the bottoms of many wetlands, in particular isolated and pothole wetlands which are not periodically flushed by flood waters. This may result in a perched water level in the wetland and little groundwater interchange during much of the year. However, this impermeable layer may be confined to only one portion of a wetland. Nearshore areas may have a smaller accumulation of organics due to the work of microorganisms, periodic drying and oxidation, the action of ice, and, in some instances, the action of waves. This zone and the bank area immediately above the wetland may serve as a recharge zone during time of snowmelt, rainfall, or flood although the wetland has relatively limited recharge potential the rest of the year.

It is often assumed that groundwater recharge is an important characteristic or function for a wetland but that discharge is not. However, discharge of a steady source of clear, cool ground water into a wetland may give it special habitat value for prized species of fish such as trout. Loucks suggests in a recent paper (Loucks, 1988) that discharge into a wetland may be important to the longevity of a wetland since discharge may facilitate breakdown of organic matter which would otherwise cause filling of the wetland.

Filling a wetland that serves as a discharge area may result in flood losses as the result of increased groundwater levels in the area and the flooding of basements. The filled area may be permanently wet, making it unsuitable for foundations, septic tank absorption fields, and other activities.

14. CAN THE FLOOD HAZARD REDUCTION VALUES OF WETLANDS BE QUANTITATIVELY CALCULATED?

Flood hazard reduction is a commonly cited wetland function or value. There is strong evidence that wetlands do, in many but not all instances, reduce flood hazards at particular downstream sites. And, flood hazard reduction functions and values can be calculated, although often not easily or accurately. The existing data base and analytical techniques for analysis of flood hazard reduction potential is much better

than that for groundwater/wetland relationships.

Wetlands may reduce flood hazards in several principal ways. Wetlands can:

- facilitate the passage of flood waters from upstream to downstream sites (flood conveyance), and
- temporarily store flood waters (flood storage).

Flood conveyance. Flood conveyance relates to the ability of a particular portion of river or stream to convey a given amount of water from upstream to downstream within a certain period of time. In general, the wider and deeper the stream or river channel and the adjacent area (including wetlands acting to convey flows) the greater the conveyance capacity. The greater the conveyance capacity, the lower the flood elevations along the reach of stream and the lower the velocities. This means less flood and erosion damage for structures and other activities carried out on the adjacent flood plain.

It is ironic that flood conveyance is perhaps the riverine wetland function most amenable to quantitative evaluation, the function for which most data exists, and the function best accepted by the courts for the purpose of flood hazard reduction regulation (Kusler, 1969, 1971, 1983). But it is rarely considered by wetland managers and is generally combined with flood storage if considered at all. An examination of riverine floodplain maps reveals that many (but not all) riverine wetlands exist within flood conveyance areas and help to convey flood flows. In some instances, wetland vegetation (e.g., heavy growth of trees) may decrease floodway conveyance but have much less impact than fill, levees or other solid structures (often alternatives to wetlands) at the same location.

In general, the closer a wetland is (horizontally) to the channel of a stream, the greater its flood conveyance capacity. Wetlands adjacent to high gradient streams with narrow floodplains play particularly significant conveyance roles. Elevation and configuration of a wetland are also important. Backwater wetlands may have flood storage potential but little conveyance capacity.

Well-accepted and broadly applied techniques are available for calculating the flood conveyance capacity of a particular area (whether wetland or nonwetland) and the impact of fills or obstructions placed within such an area upon heights and velocities. Hydrologic models such as the Corps of Engineers HEC models have been widely applied by federal agencies (FEMA, the Army Corps of Engineers, SCS, USGS, TVA), state flood plain programs, local governments, and consulting firms to map floodways and determine the impact of proposed fills or other activities in those floodways. These models determine the impact of restriction (hypothetical or actual) at a particular point along a stream for a particular flood discharge (usually the 100 year flood discharge).

Regulations of the Federal Flood Insurance Program and most state and local floodplain regulations prohibit fills or other alterations to floodway areas which will individually or cumulatively raise flood heights on other lands more than a specified amount (e.g., no rise, 1 foot).

Flood storage. Flood storage is the most commonly cited hydrologic function of riverine, coastal and estuarine, lakeshore and isolated wetlands (unlike flood conveyance which is confined almost entirely to riverine wetlands). But, the available literature concerning wetland flood storage appears to be, on its face, somewhat contradictory. To evaluate the potential of a particular wetland to reduce flood damage one must specify the magnitude of the flood event and where damage is to be reduced. Unless this is done, results of flood storage calculations or empirical studies may appear to contradict one another. These finding are consistent, conceptually, with what is more broadly known about runoff and flood peaks and with a very large number of technical flood routing studies (numbering in the thousands) undertaken to calculate the effect of dams and reservoirs and stormwater management facilities upon particular flood peaks.

Flood storage, like recharge, is a somewhat relative term. Whenever and wherever precipitation and storm tides occur, the entire landscape (to a lesser or greater extent) stores and conveys these waters from higher to lower elevations. But the amount of storage and speed of release of the water varies greatly, depending upon a number of factors. High gradient and impermeable surfaces (e.g., roads, concrete flood channels) store relatively little water (per unit of surface area) and very efficiently and quickly convey water from higher to lower elevations. In contrast, low gradient surfaces with a high roughness coefficient and a high absorptive capacity have relatively high storage and relatively low conveyance capabilities.

Depressions in the landscape (whether wetlands or not) have high storage capacity and relatively low conveyance capabilities. On the other hand, depressions sometimes also form channels which convey water from higher to lower elevations quite effectively. The storage capacity of a depression depends upon its dimensions including its depth, width, and length; the size and configuration of its outlet; the vegetation or other impediments to flow within the depression; and the "antecedent" conditions at the time of the flood event (e.g., how much water

the ground and the depression contains). In general, water very quickly runs off a surface that is already wet.

The extent to which storage or conveyance of flood water (whether in a wetland or another man-made or natural depression) will decrease flood peaks at a particular point will depend upon the timing of all sources of flood flow reaching this point. Flood heights and velocities at a particular point along a stream also depend not only upon total flow at that point but the width, depth, and shape of the channel and adjacent floodplain. In general upstream storage decreases downstream flood peaks because temporary storage allows direct precipitation and inflow from lower areas of a watershed to be carried away before water from upper areas reaches the same point along the stream. But upstream storage can, in some instances, increase peaks by synchronizing flows.

The size and shape of the outlet of a wetland is very important in determining flood storage capacity. However the effectiveness of a particular outlet and wetland configuration in actually storing flood waters also differs depending upon the magnitude of the flood. To understand this, compare a wetland to a funnel. Slowly pour a small amount of water into a funnel. If outflow equals inflow, no water builds up in the funnel. It has no storage value. Now pour a large amount of water quickly into the funnel. The discharge capacity is exceeded and the funnel fills. The water will not be slowly released.

Similarly, a wetland with a narrow outlet may provide very limited or no storage for a modest rainfall. Water runs out as quickly as it enters. Because the wetland was wet prior to the rainfall, water may run off the wetland even more quickly than it would from an unwetted upland area of similar size.

Quite another result will occur with a 10 or 50 year flood with 4-6 inches of rapid rainfall. Now the inflow to the wetland exceeds outflow capacity and the wetland fills to the rim, storing a quantity of flood water which can be calculated by knowing the area of the wetland and the difference between normal water depths in the wetland and flood depths. At the flood peak, runoff from most upland areas in the watershed would be high since most surfaces would be thoroughly wetted, making wetland storage even more important.

Any suggestions that wetlands may have limited flood damage reduction capability based upon their poor storage capability for small-scale events must also be viewed in light of the distribution of flood losses versus the size of the flood event. In general, relatively small scale flood events cause little damage because landowners plan for such events through drainage facilities, elevation of structures, etc. But landowners often do not plan for larger scale events. So, attenuation of moderate to large flood peaks is most important (except for certain stormwater losses).

Any suggestions that natural wetlands may result in very rapid runoff of flood waters for small scale events (probably true except were wetlands are wholly or partially dry) and resulting increases in flood damages must also consider the alternatives to the natural wetlands. Most alternatives would even worsen the situation. In general, placement of drainage ditches or fills in wetlands would increase runoff. Only impoundment would be likely to decrease the speed of runoff, assuming the impounded area had some storage capacity.

NOTE: *The author has divided his work between flood loss reduction projects and wetlands and broader environmental projects since receiving a M.S. in Water Resources Management (1968) and an interdisciplinary Ph.D. Degree (1970) (Water Resources, Geology, and Law) from the University of Wisconsin. He has therefore had the opportunity to spend a great deal of time with hydrologists and geologists as well as traditional wetland managers (botanists, biologists, etc.) It has been a pleasure to work with both groups but depressing to see so little communication and so much misunderstanding between them.

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chapter two

Hydrology and Wetland Classification

Hydrogeologic Classification of Wetlands in Glaciated Regions

Garrett G. Hollands
IEP, Inc.

INTRODUCTION

Given the present trend towards wetland functional assessment, it has never been more important to understand a wetland's hydrology. Thus, in the field of wetland replication, a detailed, predictable understanding of a wetland's future water balance is the most significant factor in insuring the successful growth of wetland vegetation and the functionability of the replicated wetland for the intended values. Any method used to access the function of existing or proposed wetlands must be capable of determining wetland hydrology (Adamus, 1982; Hollands and Magee, 1986).

Wetland hydrology is a complex dynamic process that controls the occurrence of soils and vegetation (Gosselink and Turner, 1978). The Hydrology Panel of the National Wetland Values Assessment Workshop (USFWS, 1984) concluded that:

> Water is the primary and critical driving force underlying the creation and maintenance of wetlands and ... a knowledge of wetland hydrology is basic to an understanding of all wetland functions. There has been little substantive work done on the hydrology of wetlands, and we lack the knowledge needed to evaluate hydrologic processes in wetlands without careful measurements....

We cannot extrapolate from the results of a few comprehensive wetland hydrology studies to all wetlands because of the complexity and variety of hydrologic systems involved.

Various federal, state and local wetland statutes recognize numerous hydrologic functions performed by wetlands, including groundwater functions. For example, the Wisconsin Administrative Code lists groundwater as one of five watershed functions and describes it as follows: "Groundwater may discharge to a wetland, recharge from a wetland to another area, evaporate from and/or flow through a wetland." In Massachusetts, wetland groundwater functions traditionally have been viewed as the recharge value of the wetland to the underlying aquifer and the role of wetland soils in preventing polluted surface water from entering the aquifer. These and similar definitions raise many issues related to wetland groundwater functions. The most common questions facing wetland regulators are:

- Is the wetland discharging water from an aquifer to surface waterbodies or is it recharging water from such waterbodies to the underlying aquifer?
- Is the recharge from the wetland to the aquifer important to other wetlands?
- What role does groundwater play in the water balance of the wetland?
- What role does the wetland play in supplying water to the underlying and adjacent aquifers used for public and private water supplies?
- What effect does the groundwater function of a wetland have upon other wetland functions, such as flood control and wildlife habitat?

This article describes a wetland hydrologic classification which is intended to provide wetland regulators with the basic geologic data needed to begin answering some of these questions. The method is applicable to any geologic region.

Traditionally, wetlands have been classified using botanical criteria (Golet, 1972; Golet and Larson, 1973). Some attempts have been made to use geological criteria to classify wetlands in conjunction with hydrological and botanical criteria (Cowardin et al, 1977). More recently, wetland classifications have been based on geological and hydrological (hydrogeologic) criteria (Hollands and Mulica, 1978; O'Brien and Motts, 1980; Jordan, 1977; Heeley, 1973; Bolter and Verry, 1978; Novitzki, 1982).

In glaciated regions, hydrogeologic classifications of wetlands must be based upon an understanding of the regional and site-specific geologic history. Wetlands in these regions occur in hydrogeologic settings created by complex and highly variable glacial geologic processes. As compared to non-glaciated areas, glaciated regions contain more complex hydrogeology and more inland wetlands per unit acre. The hydrogeologic settings are three dimensional, dynamic, and highly complex (Winter, 1976, 1981, 1983). In reality, the vegetative wetland is only a

thin green fuzz that grows on top of and as a result of this hydrogeologic setting. Periglacial processes also must be understood since they modified or added to the glacial stratigraphy underlying the wetland.

The need for hydrogeologic wetland classification became apparent when attempting to assess the functional values of wetlands under Massachusetts wetland protection statutes (Hollands and Mulica, 1978; IEP, Inc., 1979). The glacial geology of Massachusetts has been well investigated, and 1:24,000 scale U.S. Geological Survey (USGS) surficial geologic quadrangle maps exist for most of the Commonwealth. Barnstable County (Cape Cod) is used here as a hydrogeologic classification example since it has been well-studied and was the product of work performed for the Massachusetts Department of Environmental Management (DEM). Also, the wetland hydrology is relatively uncomplex, when compared to other portions of the Commonwealth such as the central and eastern portions, where a complex history of glacial-lacustrine and glacial-fluvial morphosequences occur (Koteff, 1974).

Figure 1 illustrates the method used to develop a hydrogeologic classification for Barnstable County (IEP, 1979). A review of the location of wetlands in association with the surficial geology indicates that relatively few types of surficial geologic situations give rise to wetlands in Barnstable County. Table 1 contains the surficial geologic wetland situations that give rise to inland wetlands in Barnstable County, and Figures 2, 3, and 4 illustrate the basic hydrogeologic situations of those wetlands.

SURFACE WATER HYDROLOGIC INPUT

Wetland hydrologic characteristics were identified by reviewing topographic maps and aerial photographs and by conducting field observations. Six dominant surface hydrologic cover types (S) were found to occur:

- S.1 Open water
- S.2 Vegetated wetlands other than cranberry bog
- S.3 Cranberry bog--active
- S.4 Cranberry bog--inactive
- S.5 Perennial stream
- S.6 Ephemeral stream

Many wetlands were observed to consist of a combination of these cover types. For example, open water bodies (ponds) with fringing wooded swamps and inactive cranberry bogs were all found within a single kettle wetland in outwash deposits; they were all hydrologically connected and had the same water table elevation.

Figure 1.
Hydrogeologic wetland classification flow chart.

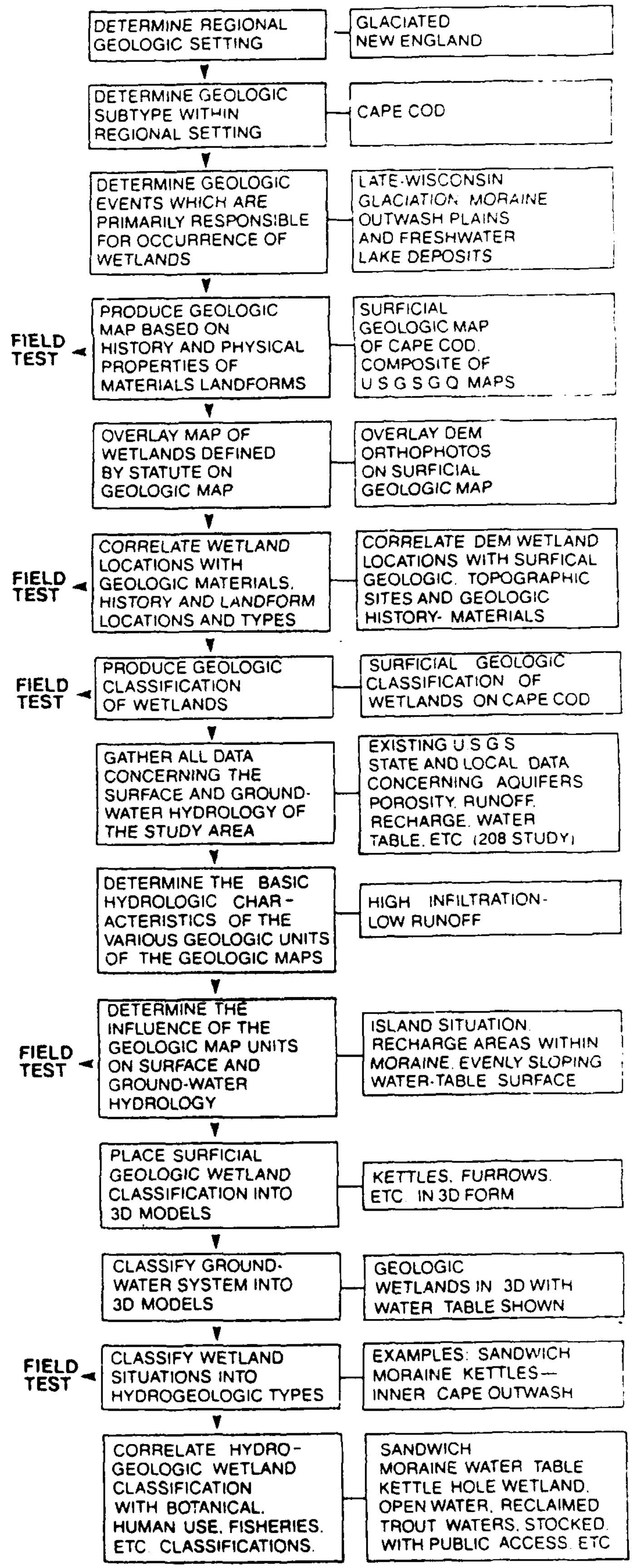

Table 1. Surficial Geologic–Inland Wetland Situations of Barnstable County, Massachusetts

OC	WETLANDS OF THE OUTER CAPE
OC1	Wetlands associated with outwash deposits
OC1.1	Kettles
OC1.2	Pamets
OC1.3	Furrows
IC	WETLANDS OF THE INNER CAPE
IC1	Wetlands associated with glacial lake deposits
IC1.1	Kettles in ice-contact deltaic-lake deposits
IC1.2	Valleys in ice-contact deltaic-lake deposits
IC1.3	Shallow depressions in glacial-lake bottom deposits
IC1.4	Shallow valleys in glacial-lake bottom deposits
IC2	Wetlands associated with Sandwich moraine deposits
IC2.1	Kettles in the moraine
IC3	Wetlands associated with outwash deposits
IC3.1	Kettles
IC3.2	Furrows
IC4	Wetlands associated with Buzzards Bay moraine deposits
IC4.1	Kettles
WO	Wetlands associated with the Wareham outwash plain
WO1	Kettles
WO2	Furrows

Hydrogeologic wetland classification also requires an understanding of the surface water budget (or inflow-outflow characteristics) of a wetland. Field observations identified four inflow-outflow surface water situations (I-O):

I-O1 Inflowing stream only

I-O2 Outflowing stream only

I-O3 Inflowing and outflowing streams

I O4 No inflowing or outflowing streams

GROUNDWATER HYDROLOGIC INPUT

A complete hydrogeologic wetland classification system must be able to recognize and define the wetland's groundwater hydrology. A review of USGS groundwater investigations of Barnstable County indicated that three groundwater situations (G) may occur:

G.1 Discharge dominated wetland

G.2 Recharge dominated wetland

G.3 Recharge and discharge occurs in the wetland

Wetlands predominantly are discharge areas, especially in a hydrogeologic situation such as exists on Barnstable County. Only under very limited and specific hydrogeologic conditions do wetland occur as recharge areas.

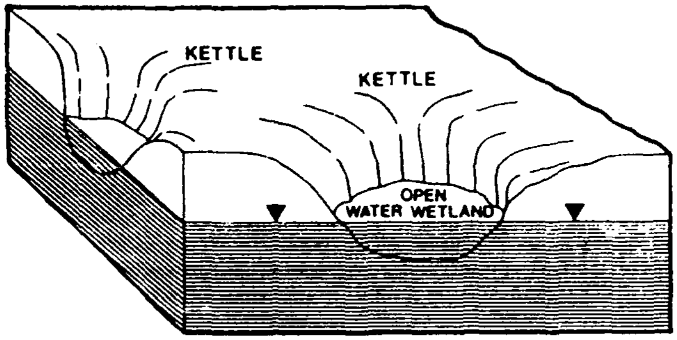

Figure 2. Kettle wetland in outwash or moraine.

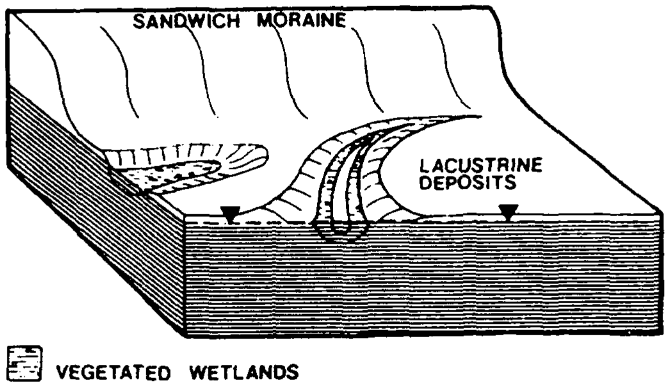

Figure 3. Shallow depressions and valleys on lacustrine deposits.

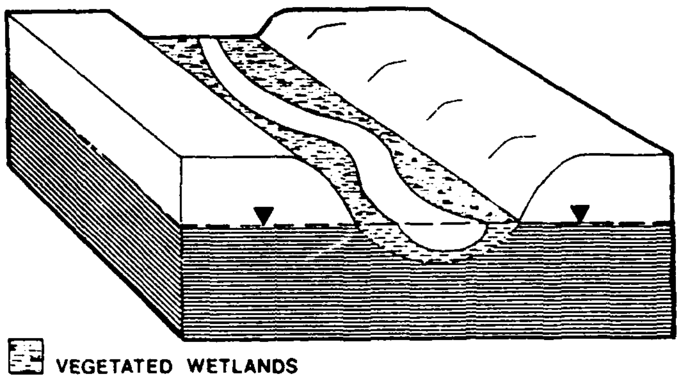

Figure 4. Furrows and pamets.

APPLICATION OF THE HYDROGEOLOGIC CLASSIFICATION

To use the hydrogeologic classification system, four steps are necessary:

(1) determine the surficial geologic wetland situation (OC);

(2) determine the hydrologic cover type (S);

(3) determine the surface water budget type (I-O); and

(4) determine the groundwater situation (G).

For example, an open water kettle pond found in the outwash deposits of the Outer Cape which has an outflowing stream and is a groundwater discharge situation would be identified using the following four-element code:

[(OC1) (S.1) (I.02) (G.1)]

If an active small cranberry bog was found in the kettle and was hydrologically connected to the larger pond, the hydrogeologic classification code would be:

[(OC1) (S.1-S.3) (I.O2) (G.1)]

with (S.1-S.3) indicating that the open water pond is larger in size than the active cranberry bog because the code number of open water (S.1) precedes the code number for active cranberry bog (S.3).

This hydrogeologic wetland classification attempts to examine the hydrogeology of a wetland in a more detailed manner than is possible using the wetland value assessment methodologies. Also, it is designed to produce an understanding of the hydrogeologic elements of a wetland. Since those elements give rise to site-specific wetland functional values, e.g., pollution abatement, groundwater supply, and fisheries, the classification can provide information critical to the application of wetland assessment techniques.

The classification system illustrates the complexity of even a relatively "uncomplex" geologic setting, such as Cape Cod. Much more complex hydrogeologic settings are commonplace in glaciated regions. Thus, the system should be applied only in the context of broad regional planning studies; it is not intended to be used on a site-specific basis, other than as the first step in data collection. Detailed site-specific hydrogeologic analysis of wetlands must involve costly, detailed, long-term hydrogeologic investigations that include, at a minimum, borings and their geologic logs, observation wells and their measurements of the water table, groundwater chemistry, and complex groundwater computer modeling (Hollands, 1986).

*This paper is reprinted from an article which appeared in the *National Wetlands Newsletter*, Mar-Apr., 1987.

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A Multi-Purpose Wetland Characterization Procedure, Featuring the Hydroperiod

James Hall Zimmerman
Department of Landscape Architecture and
Institute of Environmental Studies
University of Wisconsin-Madison

INTRODUCTION

Wetlands include a diversity of unique places where you can get your feet wet but can't swim, they are the gradient between dry land and open waters (Figure 1). Most classifications of wetlands are based on convenience (such as vegetation life form, as discernable from aerial photography) with no interpretation of causes or consequences. However, effective use regulation, management, creation, and mitigation require yardsticks for potential services and actual performance.

Van der Valk, 1982). Vegetation cycles, and even interactions with muskrats, beaver, and water birds, are driven by the hydroperiod. Hence the hydroperiod is the logical basis for explaining and predicting vegetation and wetland potential, and for relating performance to the geographic setting and impacts thereon.

This paper is not a review of the literature. Rather, it is a synthesis of personal observations in the Upper Midwest and of current wetland theory. It sketches out a model for wetland characterization and evaluation, highlighting the

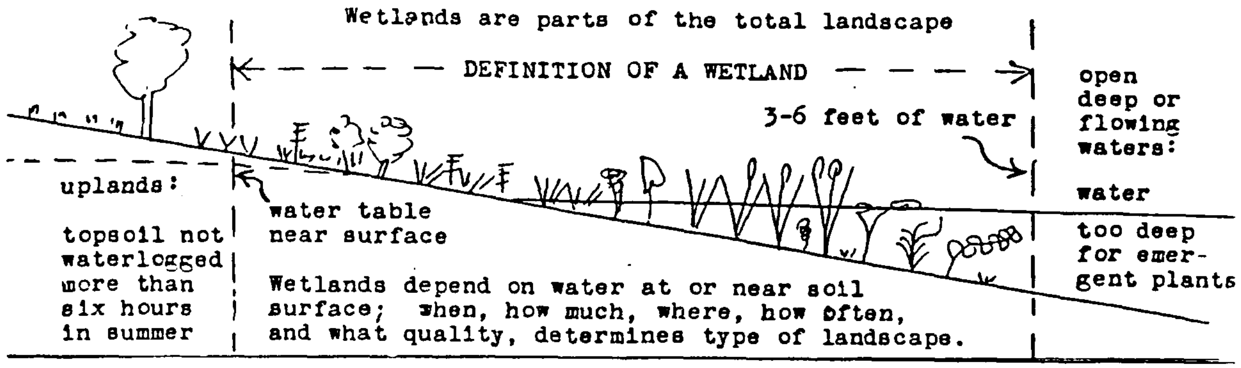


Figure 1. Definition of a Wetland

Four common misunderstandings about wetlands point to four principles of applied ecology which can form a basis for wetland characterization for many purposes (Table I). In all cases, water is the dominant environmental influence. Therefore, water should be central to the classification, delineation, and evaluation of wetlands. However, averages and extremes of water levels are too simplistic.

Instead, the dual role of water as necessary for, yet stressful to, life makes the timing and duration of its presence (the hydroperiod) the key to wetland characterization. Current wetland theory emphasizes allogenic over autogenic influences (Mitsch and Gosselink, 1986, following

role of the annual hydroperiod, in the hope of stimulating research toward refinement and application.

VEGETATION BEHAVIOR

Vegetation is the most conspicuous and enduring landscape element, yet it is changeable in response to environmental influences. We have been relying on vegetation to evaluate many wetland services, from esthetics to habitats, and as a convenient handle for mapping, classification, and boundary definition of wetlands. However, successful interpretation of vegetation must be with an understanding of its

TABLE 1
FOUR ELEMENTS IN WETLAND CHARACTERIZATION REVEALED BY COMMON MISCONCEPTIONS

Key Word	Cause	Common Misconception	Principle
System Type (Biotic and Hydrologic)	Regional & Local Biogeoclimatic Influences	All wetlands can perform the same functions and services; so they are interchangeable	On any scale, every site or system is unique; its character and potential shaped by local, regional, and historic factors.
Successional Phase (Plant Life Form and Associated Animals)	Catastrophic and Biotic Interactions and time lapse	Wetlands are transitory stages from water to dry land; so maintenance is futile	Ecosystems renew themselves through cyclic change; all phases are ephemeral, and so is peat.
Setting (Topography, Substrate, and Water Budget)	Landscape Patterns and Interactions Upstream and Downstream	Wetlands can function even if they alone are protected; so upland use and management have no responsibility	No ecosystem can stand alone, least of all wetlands, which depend on hydrologic context.
Condition or Health (Degree of potential functions realized)	Biotic Adaptations and Impact Responses	Wetlands can function even if abused or stressed; or if not functioning they may be written off as valueless	Natural and human impacts are disruptive if the biota have insufficient opportunity to adjust and recover. But impacts are reversible or may be compensated for.

peculiar requirements and behavior. Simple averages or extremes of water levels will not correlate well with either plant species or lifeforms. There is a historical reason for this.

The Wetland Paradox

To judge from their aerial flowers, most species of wetland plants (including water weeds) appear to have descended from upland ancestors. Hence they can be expected to carry adaptive mechanisms best suited to land life, where water availability is often a limiting factor. Adaptations include a reliance on energy-intensive mechanisms to absorb, pump, transport and conserve water. Etherington (1983) calls attention to the wetland paradox: in the very place where we expect water, nutrients and light (assuming clear shallow water) to be most abundant and dependable, a fourth requirement for life may be in short supply - free oxygen. Just as aquatic animals may suffer summer or winter die-offs from temporary anoxia, so may emergent plants be unable to respire. The movement of free oxygen in waterlogged soil or stagnant water can be too slow for the growth, function and maintenance of their root tissues (and other submerged parts). Moreover, feedback effects exacerbate the stress; anoxic conditions favor the reduction of metals like iron to toxic forms while fostering the equally toxic organic by-products of anaerobic respiration. Furthermore, oxygen deficiency drastically slows down the rate of bacterial decay, allowing peat to accumulate. Peat may further hinder circulation and drainage, while locking up nutrients needed by both roots and micro-organisms.

The Biomass/Waterlogging Tolerance Rule

Evidence for the stress on vegetation is the observed fact that in general the plants with the largest biomass per unit area of soil-water surface (and hence relatively large evapotranspiration surfaces as well) suffer the most in wetlands. The growth of wetland trees improves after drainage, while dams erected by beaver or man often kill even upland trees outside of the wetland by raising the water table by as little as a few centimeters in summer. In Upper Midwest swamps, at least, one may say that trees literally enter wetlands on tiptoe. They establish and persist on fallen logs and stumps that protrude above water, and the invaders in turn create new tree habitat when storms uproot them. Wetland trees tolerate wet mineral soil only in alluvial floodplains and on natural levees which, like the tip-ups of woody peat in swamps, are immersed only briefly except during the early spring floods when respiration and evaporation rates are low.

Our first approximation, then, is the biomass/waterlogging-tolerance rule. Plants may be grouped by life form according to their endurance of root-zone anoxia. Table II lists some groups for the Upper Midwest, in which the number "6" appears often enough to serve as a very rough mnemonic. Reducing their biomass with air spaces, emergent macrophytes (including wiregrass sedge, Carex lasiocarpa), have the most aerenchyma, which may serve as floats as well as snorkels. Buoyancy minimizes the need for costly (in oxygen demand as well as anabolic energy) fibrous support tissue, while providing emergency lifeboat service (floatup capacity in high water). Somewoody species do venture into deep permanent water with the facultative addition of secondary cortical aerenchyma (Taxodium, Decodon, Lythrum salicaria), or with adventitious replacement roots near the water surface (Salix; Cephalanthus). The latter shrubs and trees, at least, are thereby vulnerable to sudden drydown.

Because plants exhibit a diversity of survival strategies, the external environment - stressing the Gleasonian individualistic model of plant association and succession - fits wetlands better than does the older Clementsian one (Van der Valk, 1982). Bedfellow species may (but need not) interact. However, we find that it is floras rather than life form that tend to be sorted out in Table II when all species are included. For example, the herbaceous nettles (Urtica and Laportea) are as sensitive to prolonged summer flooding as are silver maple (Acer saccharinum) and green ash (Fraxinus lanceolata). This is our second approximation, leading toward a bioregional classification of wetland systems (see later; Table VII).

The biomass rule for species separation applies to adult individuals; for developmental stages the relationship is reversed. That is because the larger a plant becomes, the less likely is a flood to drown it, since less of the plant is submersed. Notice also that as the adaptations for inundation become more extreme (SAM's and EAM's), drought tolerance dwindles - a typical adaptational tradeoff. True aquatic species die if subjected to soil conditions of field capacity, whether the plant is a robust cattail or water lily or the seed of wild rice or Potamogeton. Note also, however, that mature individuals of most species can tolerate drought as well as flooding better than can seedlings: Potamogetons may vegetate on wet mud while willow saplings will thrive when planted in upland soils, although their seeds need a lot of water to germinate. It is a fact that most wetland species get started only on the exposed wet mud of receding shores and peat floatups, regardless of their later tolerances or requirements. Finally notice that so sensitive are different species to the relationship of water level to substrate level that peatlands may easily flop from herb to shrub to tree phases and back with very small changes in the relationship (see later; Table VI).

The Environmental Screen Model

Wetlands are noteworthy for the tendency of a seemingly randomly chosen phase of vegetation, which may dominate for years, to be suddenly and unexpectedly replaced by a totally different set of species or monotype, and sometimes even of a very different life form. An example is Gardner Marsh Fen in the UW-Madison Arboretum, one acre of whose drained, burned peat had been dominated by Urtica procera continuously since a peat burn in 1936. Then, in 1968, a summer flood from new urban development caused the nettle to be replaced in three months by a monotype of Calamagrostis canadensis, which had fortuitously seeded heavily nearby after a vandal's winter fire, and which has dominated the new site for the ensuing nineteen years. The reason for these flops is that vegetation alternately plays two games: "musical chairs" and "king of the hill". Species scramble for establishment after catastrophic devegetation and the survivors form an enduring climax-like plateau until the next severe stress opens up the site for a new lottery. The vulnerability of vegetation to hydrologic influences makes wetlands more prone to the unexpected responses to stress cited by Loucks (1985) than are upland systems. One might have chosen the term "punctuated equilibrium" to characterize wetlands had it not already been usurped by students of evolution.

Examples of the individualistic nature of species' adaptive survival strategies are the dual reproductive systems seen in hummock sedge (Carex stricta, both cespitose and clonal), hog peanut (Amphicarpa bracteata, annual and perennial, cleistogamous and phasmogamous), and yellow nutsedge or chufa (Cyperus esculentus, annual with both seeds and tubers). To fit this individualism into a predictive model for wetland vegetation, Van der Valk (1982) chose the concept of the environmental screen or sieve. Adapting his model, we can envisage five environmental screens which successively eliminate species from the race for a place in the sun when the music stops, according to their requirements and tolerances.

The first screen is the site's characteristics, including geographic location, the consequent biota, and the access of different species according to chance factors such as seed bank, distance, dispersal mechanisms and agents, and weather.

The second screen is the environment immediately following devegetation, which can be total or partial. Commonly this is the drydown after flood, but it could be flooding itself, or peat drainage with or without fire, or fire with or without return of water, or it could be animal or human soil disturbance with or without water.

The first two screens determine what species start to grow. The time of year is an important aspect of both screens.

The third screen is the ensuing condition over one or several growing seasons, strongly influenced by the hydroperiod (see later).

The fourth screen is the interaction of species and environment - a feedback system or cycle, coinciding with the third screen. It includes relative growth rates, competitiveness, persistence in time (life span as well as competition endurance) of the plants, and animal influences. The latter include consumers attracted to the plants or to water regimes (e.g., selective grazing by deer or muskrat; dispersal of woody species tardily by invading mice or shrews attracted to cover; excavation by ants, crayfish or earthworms favoring or hindering species). The third and

TABLE II
SOME UPPER MIDWEST WETLAND PLANT BEHAVIORAL GROUPS

Group	Tolerance of Waterlogging[1]	Duration of Drought[2]	Typical Species in this Region
A. UPLAND VEGETATION	Six Hours[3]	Moderate to High	Black, white oaks; sugar maple; Apple; white ash;wheat, soybeans, alfalfa; upland poison ivy; most garden plants.
B. ALLUVIAL RIVER FLOODPLAIN FLORA	Six Days[4] (roughly)	Moderate to High except seedlings	Swamp white oak; silvermaple; green ash; willows; cottonwoods; Urtica, Laportea; wetland poison ivy; waahoo; prickly ash; grape; river sedges[5].
C. SHORE ANNUALS (Moist soil pioneers)	Three to Six Weeks[6]	High	Bidens cernuua; Polygonum lapathifolium; Alisma; Leersia; Echinochloa; Panicum capillare; Cyperus esculentus and erythrorhizos.
D. WET CARR SHRUBS AND PEAT SWAMP TREES	Six Weeks[7] (but may delay damage for 1-2 yrs)	High to Low	Pussywillows; red osier dogwood; alder; poison sumac; Ilex; Aronia; bog ericads w/o floatup (e.g. Chamaedaphne); N. white cedar (Thuja); tamarack; black spruce; black ash; red maple.
E. PEATY SEDGE MEADOW, FEN, BOG HERBS	Six Weeks[8] (but longer if they float up as a mat)	Probably Low	Carex lasiocarpa, stricta, aquatilis, rostrata, atherodes, lacustris, oligosperma, buxbaumii, sartwellii; Cladium, Calamagrostis; Triglochin; Lycopus; Viola; Rumex brittanicus; Asclepias incarnata; Dryopteris thelypteris; Scutellaria.
F. EMERGENT AQUATIC MACROPHYTES (EAM'S)	Permanent Water to 3-6 feet deep[9] (but most don't need flooding)	Low to None	Bulrushes (Scirpus); cattails (Typha); Sparganium eurycarpum; Sagittaria; Pontederia; Glyceria; Nymphaea; Nuphar; Nelumbo; Brasenia; Zizania.
G. SUBMERGED AQUATIC MACROPHYTES (SAM'S) OR WATERWEEDS)	Permanent Water w/o much EAM (which would obstruct light)	Low to None	Potamogeton; Myriophyllum; Ceratophyllum; Utricularia; Elodea; Najas, Vallisneria; aquatic Polygonum & Ranunculus spp.
H. DUCKWEEDS (small floating plants)	Permanent or Semi-permanent Water with low wind access	Poor	Lemna; Spirodela; Wolffia; Azolla; Riccia; Ricciocarpus.

1. Maximum duration of summer waterlogging (once in a year) of the root zone without damage or death.

2. Soil at field capacity; effect on plants or seeds.

3. SCS specifications for drainage of terraced fields to spare crops from damage.

4. Southward, where number of trees in wetlands becomes larger, many studies have shown that tree species may be grouped according to small differences in frequency and duration of flooding.

5. In Wisconsin, Carex grayi, muskingumensis, alopecoidea, emoryi, laeviconica, davisii, and lurida are confined to floodplains for some reason.

6. Including floodplain pioneers and sedge meadow "climax opportunists" (Zimmermann, 1983). Some, like Bidens coronata, and Gerardia spp., are confined to wet sand or peat. This group (mostly short-day species) is geared to germinate any time on exposed soil but flower in autumn normally, or in early spring in very dry years when they can start in March. Some species, like Panicum and Polygonum, may also invade upland fields. All may endure permanent flooding after reaching flowering stage.

7. Assuming no floatup. Most wetland shrubs will endure flooding for several years, growing poorly or half dead, but they become vulnerable to sudden drydowns (even sandbar willow clones). Cephalanthus may endure more permanent water in our region.

8. Often able to spread toward shores by vegetative means, or to float up and persist as floating mats with high water if no wave action. They succumb if the tops are inundated, grazed by muskrats, or covered with debris or dead duckweed or algae after floods recede. Warm-season prairie species (e.g., Andropogon spp.) may occur in sedge meadows and fens, perhaps because anthills and summer daily lows in water table allow their roots to "breathe".

9. There are differences in depth tolerances. Pontederia and some Sagittarias endure high water by becoming benthic rosettes like Vallisneria, thereby being vulnerable to turbidity as is the annual Zizania. It appears that sudden alteration of waterlevel, and/or siltation, may stress or kill cattails.

fourth screens determine which plant species become dominant in the plateau, or temporary climax, phase, which is composed of those which can endure best and keep invading species out. It may be complex in species and even life form, such as the floodplain forest, or even the prairie fen peatland (Zimmerman, 1983). However, it often is nearly monotypic (e.g., white cedar swamp; cattail or bulrush marsh; Carex stricta or C. lasiocarpa meadow; or leatherleaf (Chamaedaphne) bog). The dominant species can be said to be playing King of the Hill. They are characterized by long life, often vegetative propagation, and resistance to moderate hydrologic and consumer stress. Quite possibly they have allelopathic (chemical deterrent) effects on the germination and growth of other plant species.

The fifth and final screen is the next devegetative catastrophe that starts the games again.

Table III, based on personal observations, predicts the dominance of different plant groups from Table II when given different sets of environmental screens (especially screens 2 and 3).

Resistance To Change

One further element in the vegetation model is the tendency of any living thing to endure stress and even alter the environment in its favor. Woody plants, with large evapotranspiration capability, may lower the summer water table to their benefit, as well as intercepting light needed by herbs. Shrubs, once established, may attract rodents and birds whose dung promotes woody growth thenceforth. Wetland trees may recover from a single week of summer flooding, while wetland shrubs may endure a year or two of root submersion, if no other stresses occur at the same time. It may take alternation of flood and drought, or possibly fire in combination with parasite

TABLE III
PLANT RESPONSES TO DIFFERENT HYDROPERIODS

Predicted performance, according to behavioral signatures, of some Wisconsin plant species groups under different waterlevel regimes (environmental screens 2-5) following a drydown (from personal observation)[1]

Plant Group (Table II)	Initial Drydown: Dry Soil	Initial Drydown: Wet Soil[2]	4-8 Wks of Flood 1-12" Spring/Fall[3]	1-6 days of Flood 1-12" Summer once[3,4]	1-6 Wks of Flood 1-12" Summer[3,4]	Flood 4"-3' All year	Flood over 6' deep all year
UPLAND SPP	G	(G)	P	P	P	P	P
SWAMP TREES	(G)	G	MCa	MCa	P	P	P
SWAMP SHRUBS	(G)	G	MCTabc	MCTabc	LFc	PL	P
SEDGE MEADOW PERENNIAL HERBS	D	G	MCTbc	MCTbc	LFc	PL	P
SHORE ANNUALS	(G)	G	MTabcdf	MTabcdf	MTFabcdf	MTF	FPD
PERENNIAL EAM'S	D or P	G	MCd	MCd	MCd	MCd	P or D
SAM'S	P	(G)	V or M	V or P	V, M or P	MCde	MCf
WILD RICE	P	(G)	(MC)d	(MC)d	MCd	Mcd	GP
DUCKWEEDS	P or D	(G)	MCf	(G)f	MCdf	MCde	MCe, P or D
ALGAE	D	(G)	MCf	(MC)f	MCdf	MCde	MCef

Legend: For seeds or propagules: G - germinate; D - remain dormant; P - perish. ( ) - sometimes: few species or individuals. For plants: M - grow to maturity; C - may dominate in a climax or stable vegetation; T - temporary dominance; F - may endure by floating up; L - lame duck persistence in unstable state; P - perish; V - vegetative growth. For recognized wetland type (see Table IV): a - floodplain or peatswamp forest; b - shrub carr; c - sedge meadow including bog and fen extremes; d - deepwater marsh with moist soil variants if not permanently flooded; e - open shallow to deep water (pond, stream, lake, bay); f - temporary or semi-permanent pond.

1. Waterlevel regime (hydroperiod) typical of one to several years following drydown without change.
2. Including exposed shoreline with water decline, or alluvium exposed after flood, or floatup peat mat.
3. Soil moist all through the year.
4. Also in Spring and Fall.

attack, or deer or rabbit browsing, to eliminate them. Once dominant, woody plants in wetlands resist fire which might otherwise set them back, whereas sedges and grasses tend to be very flammable, but may also resprout into especially lush and dense sods after fire or light grazing, evidently benefiting from the ensuing heat and light and nutrient availability, to the disadvantage of potential woody seedling invasions. Nutrient lockup in sedge and moss peats may be an important factor in the resistance of open phases of fens and bogs to shrub and tree invasion. Given renewed nutrient supply by floods, on the other hand, the rapid and early spring growth of monotypes of unwelcome invaders is spectacular. Examples are Phalaris, Phragmites, Lythrum salicaria, Salix interior, Urtica procera, and even the annual Ambrosia trifida.

In the wet marsh, competition for nutrients and light is keen between all five layers of vegetation: EAM's, SAM's, duckweeds, algae, and shore annuals; and even between waterlilies and either waterweeds or other EAM's. We become accustomed to the persisting phase, only to be jolted by a sudden and unexpected flop to another mode when conditions finally tip the balance to a new group. Floatups of cattails, sedge mats, and moss bogs, occasionally with young forest on them, can prolong the life of vegetation expected to succumb to higher water levels. We have, then, a number of situations in which vegetation has the status of the lame duck, obscuring for a time the environmental change underway or already in place. The lag in response has been termed "hysteresis", a term borrowed from radio tuning terminology by Dr. Calvin DeWitt (pers. comm.). Looking for signs of stress in lame duck vegetation can be an important research challenge.

Table IV documents some responses, including endurance, of one group of species (Typha) to environmental challenges, illustrating the interplay of environmental screens and

TABLE IV
SOME OBSERVED RESPONSES OF CATTAIL (TYPHA SPP.) IN SOUTHERN WISCONSIN TO DIFFERENT COMBINATIONS OF ENVIRONMENTAL SCREENS
(All in Dane County except Red Cedar Lake, Jefferson County)

Wetland Sites (Types) Dates	Cattail Response (See Text)	Apparent Cause	Comments and Explanations
S. Waubesa, Upper Mud Lake, and Lodi (mires) 1840.	Invasion and Persistence to 1987 as co-dominant in fen/ sedge meadows.	Agricultural FMC syndrome caused loading of silt, fertilizer, and mild riverization of hydroperiod.	Lodi peat cores suggest coincident increase of Typha and Ambrosia pollen at settlement horizon. Water maintained near peat surface has excluded woody invasions.
Gardner, at UW Arboretum on L. Wingra (fen mire) 1912, 1940.	Two invasions on peat-marl spoils. Monotypes persisting to 1987.	Dredging spoils provided unvegetated seedbed. Lowering water level excluded muskrats, while GWD maintained wet surface for summer survival.	Total removal of Carex allowed Typha to become fully dominant. Shrub invasion on drier spoils has not spread to cattail areas due to wetness and cattail cover.
Dunn's at Madison (deepwater marsh) 1970's.	Complete die-off in water; invasion of bordering sedge meadow.	Higher summer water levels, along with heavy siltation, and no further drydowns, as watershed urbanized around perched marsh.	Water remained 0.3-2.0m deep after 1969 drydown had allowed cattail renewal throughout marsh. In 1976 drought, a single 0.5" July rain kept basin full. Invasion of sedge by cattail due to riverization effect.
"1918" marsh. University Bay on Lake Mendota (shallow marsh) 1969-1987.	Three 2-year boom/bust cycles of invasions and eatouts. No eatout in 1986-1987.	Until leaky dam lowered water in 1986-87, muskrat buildups followed revegetation in summer drydowns of perched marsh.	Roundstem bulrush (Scirpus validus) co-dominated each cycle as incomplete drydowns favored germination of benthic bulrush seeds which do not require temperature fluctuations. This is an experimental, restored wetland.
Fish Lake Bays (deep marsh) 1970's.	Die-off with and without floatups; no return.	Persistence of high water for over 10 years, and wave action, in former shallow bays.	Deep lake, fed by GWD from large glacial moraine, has slow hydrologic cycle immune to brief droughts like 1976. Drowned shore trees were 60 years old.
Red Cedar Lake (deep marsh) (1970's.	Extensive cattail floatups have persisted to 1987.	Wetter seasons and ditch-clogging elevated summer levels somewhat in this shallow GWD-fed lake.	Extensive floating wiregrass fens and bogs in bays may have provided potentially buoyant peat near to specific gravity of water. Peat floatups without vegetation occur in some summers, sinking in autumn.

hysteresis. It also demonstrates how human impacts as much as a mile away may influence wetlands unintentionally and unexpectedly. This may be termed the upland-lowland connection (points 3 and 4, Table I; Zimmerman, 1982).

In summary, two salient points about plant behavior are: (1) that owing to adaptive radiation in evolution, vegetation response is diverse and extremely sensitive to subtle differences in hydrologic as well as associated chemical conditions; and, (2) that owing to the cumulative impact of anoxia, this sensitivity to hydrologic effects is chiefly to the duration and timing of waterlogging when plants are not dormant.

WATER BEHAVIOR

Wetland hydrology is as complex as its vegetation. Water has three physical states, and varies additionally in depth, temperature, solute content, flow rate, and in the timing and frequency of changes in all parameters. All of these features affect life and are affected by life. A logical starting place for analysis is to use vegetation response as the basis for asking hydrologic questions. Since the hydrologic regime (hydroperiod) seems to be critical for many plant species, we can ask: What are the factors affecting the hydroperiod? First, the water budget must be seen as a dynamic equilibrium between input and output rates (Figure II).

Water outputs are relatively easy to visualize. But while SO is often via a measurable stream, CWR and even ET require research initiatives for accurate quantification. Sediment (aquatard) permeability under different water pressures (depth or head) may be determined by pressure-test probing. ET is at present estimated from upland stations which may not be representative of all or any wetlands. In a growth chamber, one might study, alone or in combination, the effects on ET of temperature, light, relative humidity, air movement, plant life form and coverage of soil or water, and degree and duration of root zone saturation and anoxia.

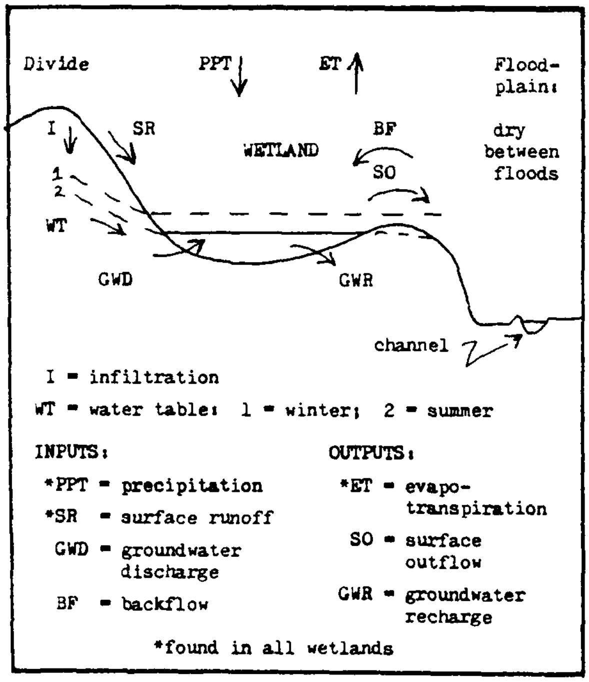


Figure 2. The Wetland Water Budget

Among inputs to wetlands, while PPT and point-source SR may be easy to measure, BF is ephemeral and may be diffuse, while CWD is difficult even to visualize. CWD is typically diffuse, may be horizontal as well as vertical, and usually varies from point to point. Wetland basins tend to have finer-textured (denser) sediments on their floors (clay, silt, marl, even peat), and coarser substrates (sand, gravel; more porous) toward shore, enabling CWD to occur most easily in spring (when WT is likely to be high enough), whereas in summer the basin may be perched over its benthic aquatard well above the receding WT below, and water depth too shallow to force water down through its floor. Hence the water tends to be dependably deep in spring, even when snowmelt is meager, and more variable in depth in summer, when it is at the mercy of weather without the stabilizing effect of base flow (CWD). Like non-point SR and SO, CWD needs some careful modeling approaches, and testing by refined piezometric instrumentation.

The Hydroperiod

In their Chapter 4, Mitsch and Gosselink (1986) clearly define hydrologic parameters and relationships and emphasize the importance of temporal pattern or regime of waterlevels (the hydroperiod) as characteristic of each kind of wetland system. This text is weak only in three types of wetlands, little understood but important to the Midwest: the sedge meadow, the groundwater slope fen, and the wet prairie (not to be confused with the deepwater marshes called prairies in the southern states). The annual (four seasons) hydroperiod is the focus of this discussion. Basically, there are five influences on annual hydroperiod which in turn affect vegetation:

1. The water level tends, in all wetlands, to be highest from fall through spring, when PPT and SR inputs are not robbed by high ET in watershed and wetland, and while therefore groundwater is recharged, raising the WT to its peak level. The peak water level is usually attained in early spring when snowmelt (SR) augments whatever CWD occurs. Except for riverine scouring, spring highs do not adversely affect vegetation, since at low tempera-

tures plant respiration rates are low, oxygen storage in water is high, and rapid flow or access to wind maximizes circulation. In early spring and late fall, Langmuir lanes downwind indicate vigorous circulation that stirs up sediments and helps cause decay for organics for nutrient recycling benefit to plants.

2. Late summer usually brings waterlevels to their lowest for the year, as cumulative effects of high ET at warm temperatures, declining GWD if it occurs, and sporadic PPT coincide with maximum water use by plants in growth and ET transpiration losses. While better access to oxygen by roots may result from shallower water, or declining water table in mineral soil or peat, the higher oxygen use rate may still pose a problem, while exclusion of wind by rampant vegetation may promote aquatic stagnation. Just as the balance between nitrogen fixation and denitrification may be easily tipped by small changes in water level and circulation, so their benefits or hazards to plant growth may be balanced on a knife edge. Studies must be fine-tuned to obtain net effects over the summer. One thing is certain: higher temperatures, combined with exposure of unvegetated peat or mineral soils, as waters recede or peat floats up, triggers germination of a host of wetland species; drydown is the classic renewer of most kinds of plant life in most wetland types (Tables III and VII).

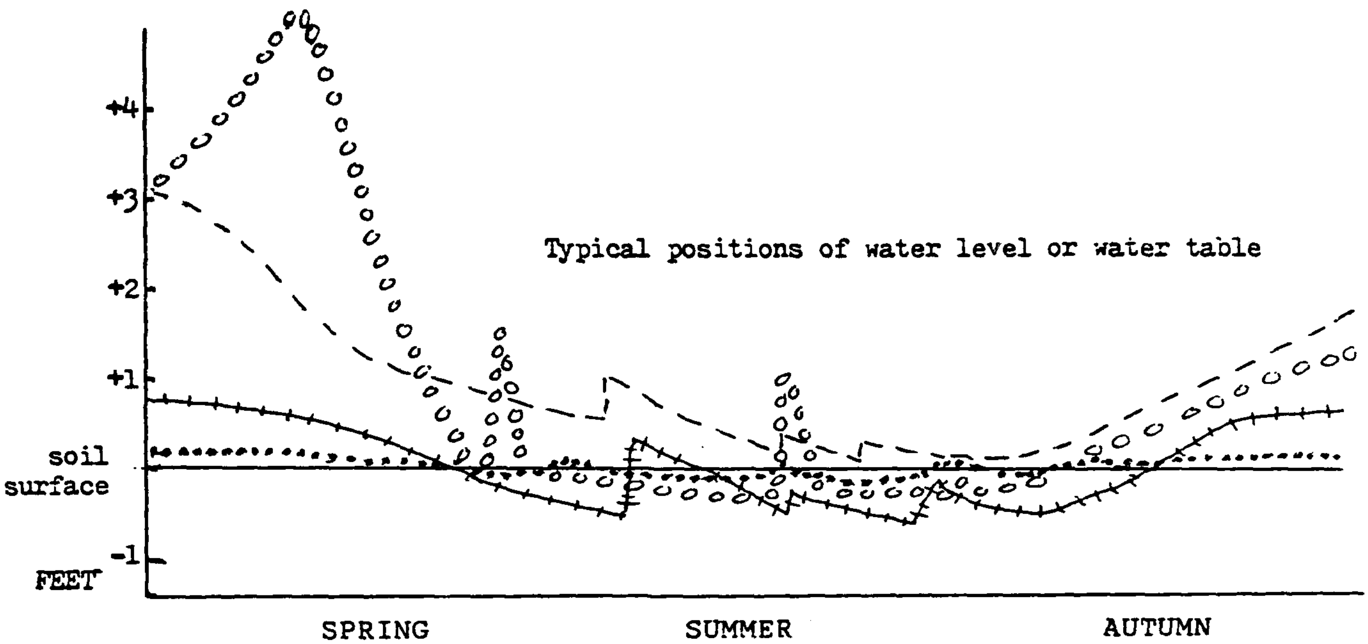
3. From their peat accumulation we may deduce that mires suffer little water level fluctuation through the year, including summer, since interruption of waterlogging would enable decay to catch up with production. Both deep water and drydown foster bacterial action, while prolonged drying out might invite fire. Knowing that woody species are very sensitive to waterlogging in summer, we may expect that small differences in water level with respectto peat surface may be critical for the vegetation of mires; their phases may thus be on a knife edge in summer. There also may be a fine balance to be tipped in favor of either vascular plants or bacteria of decay. Since bacterial respiration is so rapid in comparison to that of green plants, a slight daily summer drop in water table due to ET may enable vegetation to obtain oxygen and some nutrients from decay and/or fixation while bacterial decay is proportionally more severely hindered, except at the very surface. That disparity might explain how peat may accumulate even in cold acid bogs whose net primary productivity is as low as that of deserts (Etherington, 1983). This is a difficult realm of research.

4. At the opposite extreme from the mires is the high amplitude and erratic pattern of water level fluctuation during any season in the following two situations: (1) the high ratio of watershed surface area to wetland area characterized by riverine wetlands, which magnifies the PPT input many times; (2) the small shallow perched basins in relatively droughty climates which include vernal pools, semi-permanent or temporary ponds, and playas. Here the lack of much watershed and the lack of much stored volume make the basins very vulnerable to the vicissitudes of the weather, especially in summer.

5. Wetlands with significant groundwater input are the least affected by current weather, although they may change drastically over the years as long-term weather cycles have their effect on general area water tables. An example is Fish Lake (Table IV), whose water table rose to kill shore trees over 60 years old. When water levels seem out of phase with current weather, one may suspect GWD input. Groundwater-fed wetlands are characteristically sited in porous substrates such as gravel and sand. They often have elevated "water tanks" near them in the form of adjacent hills of sandstone or glacial ice-contact deposits such as moraines or drumlins. GWD has the effect of causing cool microclimates in warmer regions by maintaining relatively even and cool water levels throughout the seasons; hence they tend to produce peatlands similar to those of more northern regions having higher P/E ratios. However, the peat type differs, since GWD tends to maintain high nutrient levels, whereas PPT tends to wash nutrients away (see later for comparing fens and bogs; figure 5).

Hydrologic Classification

Figure 3 illustrates schematically several kinds (or signatures) of hydroperiod, ranging from the least stable ones of floodplains and small perched basins to the relatively uniform ones of mires. Now we are ready to ask: What hydrologic signatures and their causes are characteristic of different kinds of vegetation? Novitzki (1979) has made a start by classifying Wisconsin Wetlands primarily by their water sources. That is reasonable since outputs are less likely to be affected by site characteristics than are inputs, given that ET isusually influenced heavily by general climate while GWR appears to be rare in our wetlands. The input sources tell us



ooo Riverine floodplain hardwood forest
--- Deepwater marsh
+++ Semi-permanent ponds
...... Mires (fens, bogs, intermediates) in any phase

Figure 3. Some typical normal annual hydroperiods in Wisconsin

much about the hydroperiod, as well as about water quality, including temperature and nutrient flow. Adapting Novitzki's system (adding the typically solely ombrotrophic acid bog on the surface-and-groundwater divide), we have five distinguishable hydrologic types (Table V, illustrated in Figure 4). Table V begins to relate Figures 3 and 4 and leads the way to a combined hydrologic and vegetational classification (Table VI).

Wisconsin is a good place to relate hydrology to vegetation due to John Curtis' exhaustive (1959) studies. He ordinated plant communities on continua of environmental factors, especially climate (temperature or water stress) and soil moisture, although his death cut short some potential applications to wetlands. Carrying his concept into wetlands with the biomass-waterlogging tolerance rule, we can relate plant communities (segments of the vegetation continuum) to the hydroperiod and other characteristics of the setting (Table V). Curtis' designations of these wetlands continuum segments are translated here as follows: Southern Wet Forest (Riverine Floodplain Hardwoods); Northern Wet Forest (Black Spruce-Tamarack Peatswamp); Northern Wet-mesic Forests (Mixed Red Maple Peatswamp and White Cedar Swamp); Alder Thicket and Shrub-Carr (Alder and Willow-Dogwood Carr or Shrub-Swamp); Northern and Southern Sedge Meadow (Sedge Meadow in Intermediate Mire); Aquatic Plant Communities (Deepwater Marsh and Semi-permanent Ponds); Wet Prairies (same name); Open Bog (Open phase of Bog Mire); Fen (Open phase of Fen Mire).

Succession Concepts in Classification

The work of Van der Valk (1982) provides a key to relating vegetation to hydroperiod. He pointed out that the succession concept is most useful if defined simply as change and that vegetation plays the two games of musical chairs and king of the hill. We can see that two characteristics of the hydroperiod make those games possible. The "normal" hydroperiod, that is, the majority of years when the hydroperiods are close to the mean for that climate and hydrologic setting, does not stress the stable vegetation phases that happen to have established under the environmental screens of those conditions; so they may persist for a long time. The "exceptional" hydroperiod, the one far from the mean, is likely to be the catastrophic cause of devegetation. The periodicity of the exceptional hydroperiod is as characteristic of the climate and setting as is the normal hydroperiod, varying from once in a century or more in the mires to several times in a single season for the temporary

TABLE V

HYDROLOGIC CLASSIFICATION OF WISCONSIN WETLANDS (Adapted from Novitzki, 1979) See Figure 3

Type Based on Water Input	Chief Water Source	Chemistry/Hydroperiod	Peat	Plant Community & Geography[1,5,6]
SURFACE WATER SLOPE (Large river valley system of natural levees and oxbows).	SR from relatively large watershed. (GWD provides base flow).	Fertile shifting alluvium; erratic hydroperiod. Soil damp but seldom flooded in summer.	No	Floodplain (Southern Wet Deciduous) Forest; more common southwestward in prairie-forest border climate; marsh or pond where water stands longer.
SURFACE WATER DEPRESSION (May be perched over an aquatard).	PPT, and some SR from small watershed.	Often fertile when nutrients recycle; hydroperiod stable to erratic.	No	Deepwater (cattail-bulrush) marsh if drydowns occasional; semi-permanent ponds (shore annuals) if drydowns frequent.
			Yes	Peatswamps in NE forest climate.[2,3]
GROUNDWATER DEPRESSION (Porous substrate and high water table).	GWD and flowthrough, as substrate is below WT or builds up to it.	Often fertile as flow renews nutrients; hydroperiod stable through seasons.	No	Stream, pond, or spring if no drydown; deepwater marsh if occasional drydowns.
			Yes	Peatswamps and floating fens (limy) or floating bogs (if acid substrate).[4]
GROUNDWATER SLOPE (Or Hanging Bog) (water forced up and out).	GWD, causing peat buildup over artesian seep or on sand or rock shore.	Fertility may be skewed by high lime accumulation; stable hydroperiod.	Yes	Sedge meadow, varying to calcareous fen if lime accumulates, or rarely to bog if peat builds up or if substrate very low in lime; in forest climate, peatswamp tending to white cedar if lime is abundant.
OMBROTROPHIC DIVIDE (Typically over surface and groundwater divide; may be perched (water moves down and out)	PPT, causing peat buildup; may raise WT into a spreading dome; SR causes a moat.	Low fertility, becoming lower; stable hydroperiod, becoming more stable.	Yes	Acid (ericaceous Sphagnum) bog, common NE in forest climate over low-fertility soil or rock, restricted southward to kettles over divides, that are also humid frost pockets.

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1. Using Curtis' (1959) division of Wisconsin by the "tension zone" into a cool, moist northeastern Lake States forest region and a warmer, more drought-prone, southwestern prairie-oak forest zone, we can distinguish climatic effects on vegetation through a combination of temperature (flora) and hydroperiod (even vs. erratic).

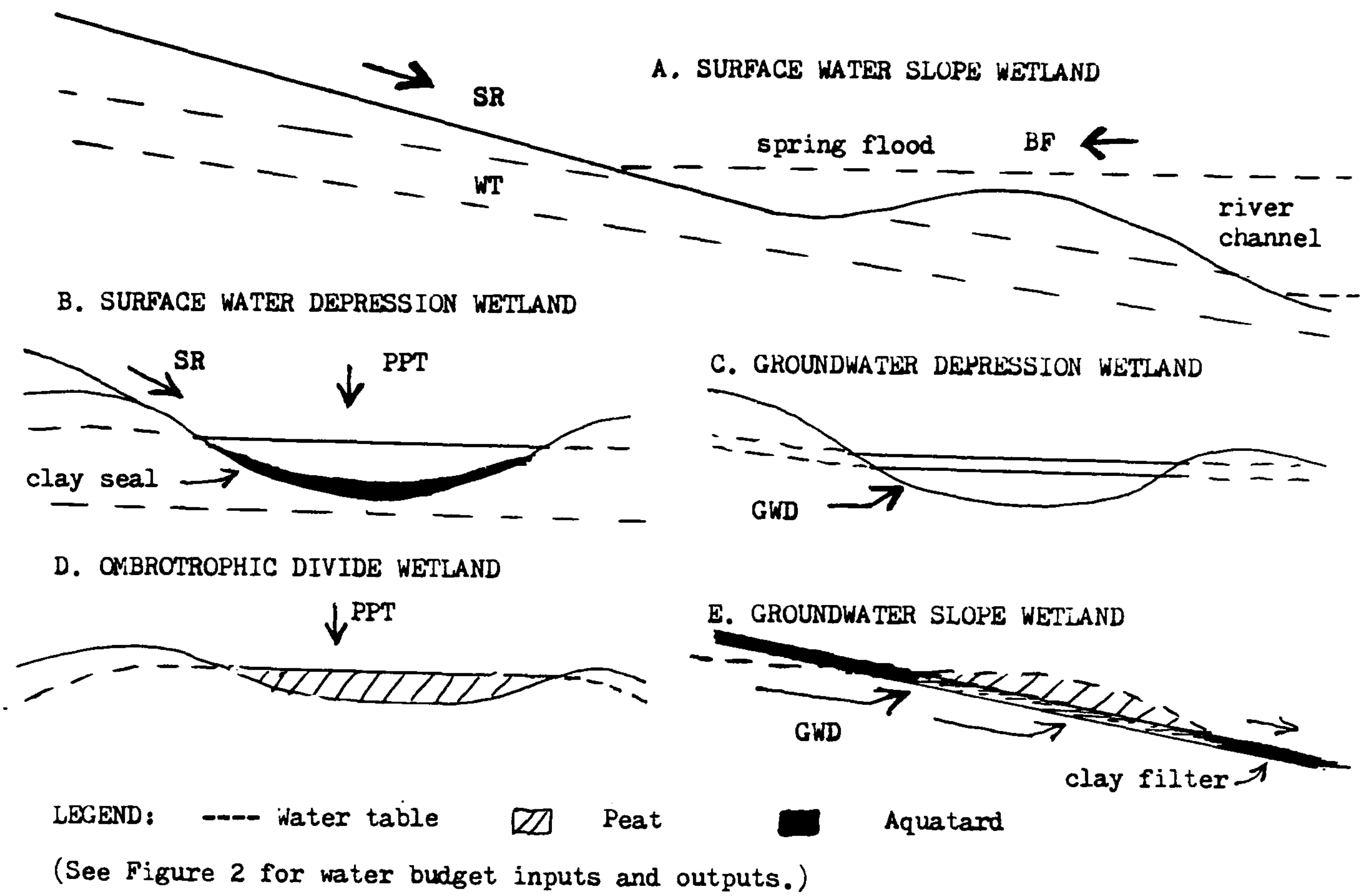
2. Although the cooler NE forest climate enables wetlands to occurr more generally there, as ET is diminished its wetlands favor trees because water levels do not often fluctuate between too dry or too wet for their existance. So trees may mature when they seed in on mossy stumps, logs, and tip-ups. These peatswamps (N. Wet Forest of Curtis) may be hardwood or mixed, or (bog, fen) coniferous.

3. Southwestward in Wisconsin, wet or low prairie may also occupy surface water depressions, perhaps requiring an intermediate frequency and timing of inundation (none severe after spring).

4. Sedge meadows often occupy groundwater depressions where water levels are low enough in summer not to submerge the plants. In the north, they recur after fire or flood removes peatswamp trees.

5. Shrubs (wet carr or shrubswamp) may occur in any type where trees also could occur, usually preceding trees in dominance owing to faster maturity if not to greater waterlogging tolerance.

6. As with vegetation types, hydrologic types form a continuum, and these are just extremes in the series.



A. **Riverine wetland fed chiefly by SR and BF from the very large surface watershed of the entire river system above this point.**
B. **Perched basin with or without surface outlet (SO), fed chiefly by PPT and/or SR.**
C. **Basin in porous substrate fed chiefly by GWD. Output may be SO or GWR as well as ET.**
D. **Basin on or near ground and surface water divide; may be perched, fed chiefly by PPT.**
E. **Site of slow persistent GWD over an artesian discharge window in the aquatard.**

Figure 4. Five Hydrologic types of wetland based on chief water inputs (after Novitzki, 1979)

ponds. While climate and setting are the usual causes of exceptional hydroperiod periodicity, animals (muskrat, beaver) and man can alter both the level of the means and the periodicity of the catastrophic conditions by digging, damming, or draining.

Van der Valk's work also helps us distinguish the two types of succession in vegetation. Type One is change due to the relative maturity rates and longevities of plant species and groups that germinate simultaneously together after catastrophe (See Table III). Type Two is change due to chance replacement in the musical chairs game: the different results when different sets of environmental screens prevail at each catastrophe. Finally, Van der Valk enables us to see the exceptional (catastrophic) hydroperiod as the chief cause of succession or renewal, while the normal hydroperiod is the cause of stable or stagnant vegetational and productivity phases between catastrophes. Table VI attempts to relate these concepts, so as to classify wetlands simultaneously by hydrologic, climatic, and geological settings, on which the successional phases are superimposed as transitory and secondary in importance.

It is in the mires that we see this effect best, where small differences in water level with respect to peat surface may make a big difference in vegetation life form. Until recently, it was thought that succession was autogenic; that is, due entirely to feedback, such as peat accumulation raising the surface, thus enabling trees to replace shrubs or herbs, or raising the water table, enabling the wetland to regress to wetter phases and to spread out. However, it is clear that peat will not build up beyond the waterlogging line, nor will it spread, without

TABLE VI
A SIMPLE CLASSIFICATION OF EIGHT WISCONSIN WETLAND TYPES
Demonstrating Vegetation Response to Hydroperiod and Water Sources

I. NON-PEATY WETLANDS (Intermittent Oxidation with Drydown or Water Movement)

A. STANDING WATER (Aquatic Species). Chemistry of soil and water varies

PERMANENT WATER	OCCASIONAL DRYDOWN (Closed Phase) OCCASIONAL FLOOD (Open Phase)	FREQUENT ERRATIC DRYDOWNS
1. SPRINGS & STREAMS	2. DEEPWATER MARSH	3. SEMI-PERMANENT PONDS (Playas)
Watercress, Waterweeds, Algae	Cattail, Bulrush, Waterlilies, Algae Duckweeds, Waterweeds, Shore Annuals	Shore Annuals, Algae

B. FLOODED IN SPRING (Occasionally briefly at other seasons). Warm summers

DAMP IN SUMMER	DRY IN SUMMER
4. FLOODPLAIN HARDWOODS	5. WET PRAIRIES
Fertile shifting alluvial soils	Organic soils from fibrous roots
Silver maple, Elm, Ash, Willow, Cottonwood, Swamp Oak, Poison Ivy, Nettle Mosquitoes	Prairie Grasses and Sedges; Prairie Forbs such as Prairie Dock; occasional shrubs like White Spirea

II. PEATLANDS OR MIRES (Oxidation hindered by constant waterlogging, low temperatures, and feedback effects from accumulating peat)

TYPE	WET (Open) PHASE	INTERMEDIATE (Carr) PHASE	DAMP (Forest) PHASE
6. FENS			
Rheotrophic, Limy Sapric Peat	Sedges, Grasses, Calcicoles	Pussy Willows, Red Dogwood, Alder, Bog Birch	White Cedar (Thuja) Tamarack
7. INTERMEDIATE MIRES			
	Sedge meadows	Willows, Dogwood, Alder	Red Maple, Elm, Ash, Birch, Pine, Fir
8. BOGS			
Ombrotrophic, Acid, Fibric Peat	Sphagnum, Sedges, Acidicoles	Heaths (Ericads), esp. Leatherleaf; Sphagnum	Black Spruce, Tamarack, Sphagnum

Notes: Any type may intergrade with any other. Continua could be documented in the style of *Curtis' Vegetation of Wisconsin,* based on the inverse relationship of plant size to flooding tolerance, and on climate and soil chemistry. It is important to note that all wetlands can go through wet and dry phases and that plant establishment is mostly during the dry phase following a die-off from flood; fire, floatup and downlogs help too. Beaver and muskrat are vey influential. Wetland types and phases are extremely sensitive to changes in water level regimes and water quality.

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climatic assistance. Instead, we can see these changes as the result of a drop or a rise in the input/output ratio in the water budget. These changes are allogenic causes; that is, the result of outside influence. Accordingly, Table VI equates herb, shrub and forest phases of the Wisconsin mires as merely interchangeable phases of the same system, and designates chemical differences as more important in distinguishing types for the purposes of functions and values useful to man. The chemical differences are likewise allogenic, a consequence (secondary effect) of the setting (water source and substrate).

If we see chemistry and hydroperiod pattern as two aspects of a wetland's characteristics, we can use them as basic causes of wetland behavior and performance, while life form of vegetation tells us which stable (temporary climax) or unstable (ephemeral) phase of the wetland we happen to encounter. The peaty wetlands (mires) all have three phases dominated respectively by herbs, shrubs, and trees (Table VI). In the non-peaty wetlands, all phases may be unstable or short-lived (deep marsh and semi-permanent ponds), or some may be short (herb and shrub) while one is long-lived (floodplain hardwoods that replace them by succession, type one). Wet prairie is poorly known, having been nearly extirpated by agriculture; but it may parallel

floodplains in having one long-lived sod phase and a very brief temporary shore annual phase following summer flood or a bison wallow. (Wet prairie also has an anomalous organic soil, due not to perpetual anoxia but to buildup of an excess of rot-resistant fibrous root accumulation, a feature shared with sedge meadows).

Table VII combines the site and successional characteristics of five Wisconsin wetlands to emphasize that the one is basic and the other superficial in classification from a system point of view. It also stresses the point that each biota has a geographic center; far from their centers the eastern pothole marshes or southern acid bogs have fewer species and are more sensitive to disruption by human impacts in the Wisconsin region.

A Wetland Continuum

We have clung to the concept of biogeoclimatic types for the sake of simplicity. Nature refuses to be categorized. The hydrologic types (Table V), hydroperiod types (Figure 3) and vegetation types (Table VI) are, of course, all idealized points along continua; and most of them are extremes. What do we get if we look at a sort of average wetland, with a balance of PPT, SR and GWD inputs and with some access to oxygen and nutrients without any added stresses, and then proceed to alter the hydroperiod and chemistry in various ways to stress the vegetation further?

Figure 5 is an attempt to relate the wetlands of Table VI to three sorts of stresses. The y axis measures the pattern of hydroperiod, from erratic in timing and amplitude (at the top) to very even through the seasons and the years (at the bottom of the page). At the top we find the semi-permanent ponds and riverine floodplains, and also the drier continental climates. At the bottom, we find the mires, respectively due to high GWD (left) and to high PPT (right) input. Linking them, the x axis represents the chemical factor. On the right is the acidic, oligotrophic, ombrotrophic or podzolic condition (downward or lateral leaching) requiring both the high P/E ratio of boreal or maritime climates and a low-nutrient substrate (or isolation from it). On the left is the high salt concentration (from movement of solutions up or out to the surface) which in our prairie-forest border climate and calcareous substrates produce basic fen mires.

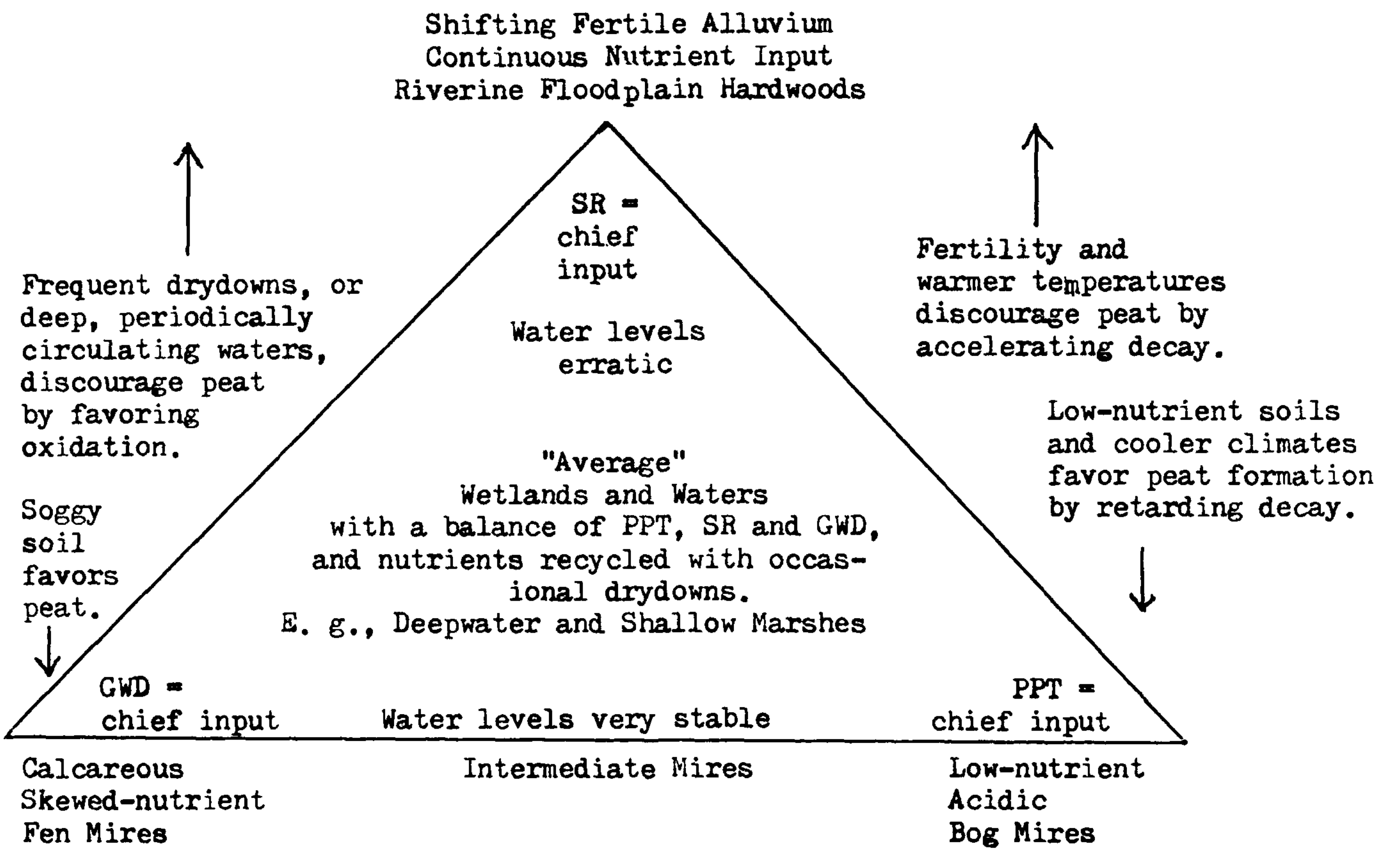

Figure 5. A tentative continuum of some Wisconsin wetlands

TABLE VII
COMBINING TYPE AND PHASE

A proposed classification of five major central-northeastern North American bio-geo-hydrodynamic wetland systems and their characteristics, based on plant requirements and tolerances. Wetlands are collectively defined as having water near or at the surface during significant periods of most years, anoxic (including organic) soils, and wetland vegetation or, better, absence of upland vegetation. However, the ecosystems included in this definition are diverse in origin, dynamics, and potential.

SYSTEM AND CLIMATIC REGION CENTER	CHARACTERISTIC SPECIES (Beaver and *(amphibians can occur in all types)*	HYDROLOGIC FEATURES OF IDEAL TYPE	SUCCESSIONAL CYCLE: Brief Phases	SUCCESSIONAL CYCLE: Enduring Phases
DEEPWATER MARSH (incl. shallow marsh and semi-perm. open ponds; related to lake and stream) "AEROTROPHIC" (Oligotrophic to eutrophic)	Cattails, bulrushes, Arrowheads, wild rice, Waterweeds, duckweed, and shore annuals; Muskrat, waterfowl; Herons, blackbirds, shorebirds, large or small fish (exc. ponds); Crustaceans, mollusks. Ponds: rare species. Assoc.: low prairie, corn, wheat	Small watershed; often groundwater input (ideally) and no siltation; high Sept.-April, low August; gradual changes; WATER 1-200cm deep (treeless gradual shorelines)	SEEDLINGS ON WET MUD → WATER LEVEL LOW ↑ HIGH ↓ FLOATUPS, DECAY	DENSE VEGETATION ↓ INTERSPERSED EMERGENTS (muskrat work) ↓ OPEN WATER AND SUB-MERGENTS → (to FLOATUPS, DECAY)
SEDGE PEAT MEADOW (Neutral to calcareous to dystrophic; anoxic) MINEROTROPHIC (esp. fen variant) (MIDWEST ECOTONES)	Tussock & other sedges, Bluejoint grass, Dogwoods, willows; Mice, raptors, snipe, cranes, bitterns, pheasant.. Rare fen plants & birds of prey. *Carr*: songbirds, woodcock, rabbit, deer	Small watershed; steady groundwater supply through year; water table near surface; no siltation	SEEDLINGS ON WET SURFACE → WATERTABLE LOW ↑ HIGH ↓ PEAT FIRE, DECAY	OPEN SEDGE MEADOW ↓ INTERSPERSION (sedge/shrub) ↓ CARR AND (Occ.) TREES
MOSS PEAT BOG (Acid, anoxix, oligotrophic, dystrophic) OMBROTROPHIC (BOREAL FOREST)	Pioneer rushes, orchids, Wiregrass sedges (mat), <u>Sphagnum mosses</u>, Insectivorous plants, Ericad carrs; Tamarack, black spruce; Special vertebrates & plants esp. northward	Low-nutrient substrate; small watershed and/or on groundwater divide; <u>rain-fed</u>; no siltation; starts as floating mat; may form moat	SEEDLINGS ON WET LOGS AND PEAT FLOATUPS → FLOOD WATER LOW ↑ HIGH ↓ FIRE DECAY, WINDTHROW	RUSHES ↓ WIREGRASS MAT ↓ MOSS CARR ↓ CONIFERS
WOODY PEAT SWAMP (Shallow, acid to calcareous, anoxic) "LIGNOTROPHIC" (NORTHEASTERN FOREST)	White cedar (calcareous, northward); or Red maple, black ash, elm, Alder, willow, birch, fir, pine. Special and general mammals and birds of forest and edges. Mosquitoes (Deer eat cedar)	Often perched water table; often small watershed; water near surface; fluctuates little; much fallen wood	SEEDLINGS ON LOGS, STUMPS FIRE, DECAY → WATER LEVEL LOW ↑ HIGH ↓ FLOOD, WINDTHROW	MUD-ALDERS OR WOOD & MOSS - WHITE CEDARS - TAMARACK AND/OR DECIDUOUS TREES
RIVER FLOODPLAIN BROAD-LEAVED FOREST (Mud or sand, eutrophic, islands and levees system) "ALLUVIOTROPHIC" (SOUTHEASTERN FOREST & MISSISSIPPI VALLEY SYSTEM) (Sloughs, oxbows - see Deep Marsh)	Silver maple dominant in Midwest; Cottonwood, willow, elm, green ash, swamp oaks; Vines nettles, ragweed; Mosquitoes, rookeries, waterbirds, raptors, upland birds, mammals. Southern plants, animals. Sand: H. locust, R. birch, abundant reptiles and mammals	<u>Large</u> surface watershed; often groundwater input or output; water levels fluctuate widely and rapidly; tolerates prolonged spring floods; heavy siltation and erosion, and low summer water table, but remaining humid and damp.	SEEDLINGS ON EXPOSED MUD → WATER LEVEL LOW ↑ HIGH ↓ FLOOD, WINDTHROW, EROSION	PIONEER HERBS AND MOST SWAMP SHRUBS AND TREES ↓ SHADE-TOLERANT DECIDUOUS TREES AND HERBS

Our fens share a few species like Triglochin maritima with the coastal salt marsh. High accumulation of Ca and Mg may skew the nutritional balance (such as tie-up of phosphates at pH 8.5), which may explain the occurrence in our calcareous fens of the same group of carnivorous and other species like tamarack and bog birch characteristic of acid bogs (except Sphagnum and the Ericads). These plants may expend so much energy obtaining nutrients from external or inadequate local sources as to be unable to compete with wetland generalist species in the intermediate (neutral pH) mires. Farther west, in the plains, the hydroperiod becomes more erratic, and we move up the y axis, where the low P/E ratio cause salts to accumulate in all wetland types not flushed by GWD.

Finally, the z axis, running vertically (perpendicular to the page), represents a continuum of mean water depth, taking Curtis' (1959) moisture continuum down into the wetlands and waters (Figure 1; Table II). Z can range from permanent standing or flowing water over six feet deep (open waters) below the page's plane up through the page's plane (water levels we see in Table VI and Figure 3) to no waterlogging at all (PPT drains out of the topsoil in less than six hours) in some plane above the page (uplands). In our midwestern climate, the three-dimensional continuum is appropriately triangular. The three points in the plane of the page (y and x axes) give us three extremes, each dominated by a single water input (GWD, PPT, and at the top, SR, which includes BF). These three points, respectively, give us our three most disparate wetlands: fen, bog and alluvial forest.

The continuum triangle is useful to relate wetland classifications originating from different perspectives. For example, what is called a fen in Maine or Northern Minnesota would be called a bog in Southern Wisconsin. The x axis provides a scale for measuring and plotting bog-like and fen-like characteristics on a single objective scale. The triangle may also help us understand temporal changes in a wetland. It can be used to plot autogenic and allogenic feedback effects such as: drying of the peat by accelerated ET when trees invade, pushing the wetland temporarily up the z axis; accumulation of peat, isolating a fen from the groundwater by absorbing rain, so it becomes more ombrotrophic (pushing the wetland temporarily to the right on the x axis); or causing a wetland's hydroperiod to become more erratic by causing an increase in SR by intensifying agricultural or urban impact (higher runoff coefficient from watershed), pushing the wetland vertically up the y axis (not irreversible if land use can be modified to control SR).

In management, we can change the wetland's phase or maintain the same one by causing or preventing catastrophic hydroper[illegible] without necessarily moving it on the continuum triangle. We could rate waterfowl productivity on all three gradients by comparing several wetlands plotted thereon, which would enable the isolation of pertinent factors such as optimum conditions between a rise in available fertility (SR input and recycling by drawdowns) and a fall in water quality, as well as optimum conditions between too much and too little variation in water level for desirable water-vegetation interspersion and muskrat interactions and control. It should be noted that management goals require maximum services at minimum costs. Stability and diversity are two such goals. The best way to achieve them is not to alter water levels artificially by dams, pumps and drains, but rather to manage microtopography once and for all and let the weather do the manipulation. If shorelines are very gradual (1-5% grades) and bottoms are encouraged to become bumpy (via muskrat activity and tree tip-ups), the small "normal" year-to-year hydroperiod variation can assure a diversity of vegetation phases and water depths within and between adjacent wetlands. For creating, enhancing and restoring wetlands, the continuum triangle will set attainable service goals on the three gradients from the position of the site.

WETLAND CHARACTERIZATION

We are now ready to create a checklist of features necessary to characterize and evaluate a given wetland for any purpose (see Table I). First, the geographic region gives us a characteristic climate, biota and geological and land use history. Second, within those constraints we have a set of biotic-hydrologic wetland system types (or extremes of continua) characteristic of certain settings (for example, Table VI). Third, the hydroperiod (with its "normal" and "exceptional" aspects) is the control feature, translating the setting into the biotic system, giving clues to the water budget which determines water quality, governing its succession phase at a given time, and sensitively indicating hydrologic impacts in the vicinity.

Finally, we need to address the wetland's potential services or values and determine to what extent these are actually being performed. Widespread human impacts, which impair wetland function, are the rule today. However, most impacts can be moderated, reversed, or compensated for. A poorly-functioning wetland should not be neglected if we have made an investment in protecting it; nor should an already impacted wetland be written off as lost when preparing impact statements and negotiating mitigation.

Wetland Potential

Each wetland, like any landscape, is unique.

Even a broad category, such as marsh or swamp, has certain general and special values. In a geographic region, it is possible to list important wetland services and then assign them quantity and quality indices for each combination of local wetland type and setting. Among self-renewing physical services, flood spreading and base-flow augmentation might be quantified on a scale of coefficients for wetland type and size, for use with a ratio of upstream watershed loading to downstream requirements. Among renewable biological services,every wetland's gene bank potential can be had from the composite inventory of that type in the region, such as the fauna of temporary ponds and the flora of bogs and fens. Production capability for fish, fur, and fowl is readily quantified from known management paradigms.

Among socio-cultural services, historic records (often non-renewable) reside in peat and other sediments of varying usefulness. Esthetics, open space, educational and recreational uses apply to most wetlands, but vary according to access by people and sensitivity to use intensity. Loadings (asking a wetland to perform new services or handle a larger quantity of water or pollutants than it had under natural conditions) require site-specific management to offset the impacts, such as removal of trapped nutrients or silt, and prevention of channelization or hydroperiod alteration by flood or wastewater flow. Careful analysis in advance will tell if such harnessing of a wetland will be both effective and economically sound, and will not compromise this or some other service the wetland performs. All these evaluations must be made in the context of supply and demand by addressing all four types of clients in a given geographic area: owner, neighbor, general public, and the unborn.

Assessment of potential is thus a crucial part of wetland characterization. It is the essential yardstick for resource-performance land use (and water use) regulation. At present, performance regulation usually provides only people-performance guidelines which too often allow regulatory laws to be circumvented or allow the letter of the law to interfere with its spirit. Just one example is the inadequacy of culverts required under roads crossing lowlands to allow for the complexity of water movement through peat. Resource-performance criteria based on site evaluation provide a true yardstick for measuring the success of restoration, creation, and mitigation. Finally, resource-performance is a measure of wetland condition that can indicate the presence of subtle unintentional impacts on natural as well as managed wetlands. Three examples follow.

Impact Analysis

Impact is defined as an unusual change in some environmental factor to which the biota is not accustomed or requires time to adjust to; the synonym is stress. Impacts fall into four categories: chemical (air or water pollution); biotic (removing or adding species, or altering species balances); topographic (changing slopes or erecting barriers); and hydrologic (altering inputs or outputs, thereby changing the sensitive hydroperiod pattern on a daily, seasonal, or year-to-year basis). Hydrologic impacts have secondary chemical and biotic impacts. For example, raising the peat level with respect to the water table by dredging or drainage can cause organic and inorganic outputs, lowering dissolved oxygen downstream, while inviting invasion of alien pests like purple loosestrife (Lythrum salicaria) or woody species upon the peat as the native flora is weakened. In wetland protection, management, restoration, enhancement, or creation, special attention must be paid to the mean and extreme hydroperiods and their frequency. Note that stress may include removal as well as addition of exceptional hydroperiods (catastrophic years) in the time sequence. An especially common impact is excess loading with water or pollutants. It is a common consequence of mistakenly expecting the wetland to buffer waters against the added impacts of intensive land use, such as agriculture, urbanization, and wastewater treatment.

Three common impact syndromes which severely hinder many types of wetlands from realizing their full potential today deserve attention since most wetland regulation is focused on the more obvious filling or drainage activities. The syndromes may be called "riverization", "peat thirst" and "lagooning".

"Riverization" is the increase in amplitude and frequency of flood peaks caused by the "flood-mud-crud" syndrome (FMC, Zimmerman, 1982), which moves marshes and meadows up on the y axis of Figure 5 by vastly increasing the SR/CWD ratio of inputs. The cause is either farming or urbanizing of the watershed or addition of sewage effluent. Loss of aquatic vegetation cover, water quality, and waterbird production are the consequences (see Dunn's Marsh, Table IV). Riverization may be detected by water quality deterioration (eutrophication, suspended solids, declining dissolved oxygen) and by a change in the biota (stress on EAM's by high floods or by carp activity; replacement of Carex by Typha or of Typha by Sparganium eurycarpum or Scirpus fluviatilis, or of Nymphaea by Nelumbo). Forested wetlands are drastically affected as well, since the slightest rise in average water level will prolong the duration of peaks in the hydroperiod (Figure 3), causing the death of trees (see Table II).

"Peat thirst" is a second, less sudden response to impact, again often unintentional. It causes fen-sedge meadow degradation by slightly

lowering the summer watertable. It can result from regional drainage (often outside of the wetland proper), or through drainage in summer by a ditch intended only to pass spring floods, or from pumping groundwater for irrigation or urban use, exacerbated by the FMC syndrome which may deposit silt buildups and/or reduce uplandgroundwater recharge. Peat thirst, in conjunction with fire stoppage (Curtis, 1959), peat fires (Vogl, 1969), and human and animal disturbance (Zimmerman, 1983) invites a variety of invasions of persistent monotypic vegetation of little wildlife value: grasses (Phalaris, Phragmites); forbs (Aster, Solidago, Urtica, Lythrum salicaria, Ambrosia trifida); shrubs (Cornus, Salix, alien Rhamnus and Lonicera) and trees (Willow, Cottonwood, Elm, Boxelder, Ash, and Aspen). Consequent changes in animal life are documented for Wingra Fen (Zimmerman, 1969).

"Lagooning" results from a desire by the owner or manager of a mire to see open water and a dream of producing waterfowl. Too often, erosion or decay of piled spoils or steep shores compromises water quality, tree or alien species invasion of spoils deters water birds and obscures the view, and the basins are too small to harbor water bird reproduction. In any case, the loss of the very gradual natural wetland edge removes most habitats and the option of biotic adjustment to waterlevel fluctuation. Filling at the upland edges of marshes and meadows has the same result, while adding a barrier (development) to upland-lowland animal migration.

RESEARCH NEEDS

Desired research initiatives range from data-gathering to testing of hypotheses:

1. Building a data bank of individual species' responses to environmental conditions (screens). Anderson's (1968) compilation of the germination requirements of native plant species could be brought up to date and extended to include survival strategies, flooding tolerances, and responses to nutrient fluxes. A simple device to test plants of different ages without altering water levels in a ditch, pond or marsh, is suggested in Figure 6. One urgent practical application is ecological control of pest species such as Lythrum salicaria. For wide-ranging species like Phragmites, races and climates may introduce variables.

2. Refining procedures to characterize water budgets. Both peizometric field work and growth chamber experimentation are needed to model GWD, GWR, and ET under varying conditions.

3. Field monitoring of vegetation responses to hydroperiod management. In particular, peatland dynamics needs attention. How does a sedge meadow originate? Why do peat mats float up (often with warmer weather as well as higher water) and why do they sometimes sink? How does the interaction of water, animals, and fire influence the successional cycle in mires?

4. Determining loading capability. The limits of wetland tolerance of uses and abuses have yet to be determined systematically in relation to type, phase, setting and condition, for various additions of floodwater, heavy metals, nutrients, road salt, pesticides, silt, and sewage.

We can seize upon the growing number of opportunities provided by environmental impact analyses and predictions, and mitigation and creation projects, to experiment (provided that we build in adequate controls and monitoring over the years) with refining hydrologic, vegetation and loading tolerance models, and facilitating classification and management.

Wetland research has suffered from the unfilled need for integration of specialist approaches, such as botany with hydrology, and management with basic ecology. Growing experience with the relationships of biogeoclimatic and setting factors to hydroperiod and vegetation may be expected to progressively eliminate the need for measuring so many environmental parameters.

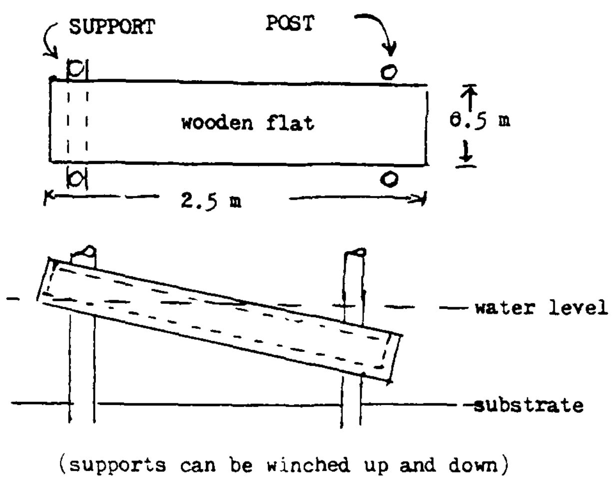

Figure 6. Plant responses to water level regimes can be tested in a flat full of soil suspended from four posts. It may be tilted, raised, or lowered to simulate hydroperiods without having to control the water level in a pond or ditch.

SUMMARY

Wetland characterization for any purpose requires consideration of the biogeoclimatic type, the local site and context, the successional phase, and the condition (degree of stress). Efficient application requires a model which uses as few indicators as possible - those which integrate many environmental influences and consequent wetland capabilities. Personal observation and current wetland theory suggest focusing on the biota (species present) and their responses (especially of plant species) to the hydroperiod (water level pattern through time, especially through the seasons). The hydroperiod is especially sensitive to the patterns of local microclimate (P/E ratio) and water budget (especially surface and groundwater inputs). Plants play the games of musical chairs and king of the hill. Successional phase turnover and duration depend largely on the matching of two signatures: the sequence of normal and exceptional hydroperiods (respectively, environmental screens 3,5) and the survival strategy patterns of locally available plant species.

Wetland management and evaluation can be assisted by locating the site on a three-dimensional continuum of wetness, hydroperiod, and soil chemistry, together with analysis of past and potential environmental impacts. Impacts (mostly human) include changes in topography, hydrology, species balances and unintentional and intentional loading. Research needs directives to refine information, as well as models for water inputs and vegetation responses, and may utilize impacts and mitigation efforts to test models for estimating wetland services and tolerances of loading.

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chapter three

Hydrologic Requirements of Wetlands

Hydrology, Disturbance, and Vegetation Change

William A. Niering
Connecticut College

INTRODUCTION

Hydrology plays a major role in controlling wetland vegetation dynamics. Although seasonal fluctuations in the hydrologic regime are a normal part of most wetland systems, it is often the unpredictable disturbances or catastrophic events that play a significant role in initiating vegetation change.

White (1979) has highlighted the role of disturbance in natural ecosystems. Wetland ecologists are also questioning the relevance of traditional succession and climax concepts in interpreting the complex of factors, both autogenic and allogenic, that are controlling vegetation dynamics (Niering, 1985; 1987; in press). Succession has been traditionally viewed as an orderly predictable and directional process leading to a climax. Over a half century ago Gates (1926) proposed a marsh-swamp to dry land sequence as the potential fate for wetlands. This was obviously an oversimplification of the process since most wetlands are not transformed into upland ecosystems; rather, they are hydrologically pulsed and have therefore persisted for thousands of years. In fact, if wetlands are doomed to become uplands, what is the rationale for legislation to protect these resources? Currently, most ecologists have a much more flexible view of succession and climax, often adopting Whittaker's (1975) climax pattern which implies a mosaic of climax types along an environmental gradient. Others, however, find these traditional terms of a limited usefulness, as noted by Elger (1947) four decades ago--

> "The term Succession, in the minds of some, appears to denote a succession of step-like metamorphoses from one association to another. Furthermore, the retrogressive-progressive argument makes it necessary for one to know whether he is 'coming' or 'going', a stand which the writer cannot always take...and which others usually settle more by faith than by empirical knowledge. The climax, and God, have certain things in common for certain botanical atheists. To paraphrase Julian Huxley, the writer does not believe in the climax, because he thinks the idea has ceased to be a useful hypothesis."

The purpose of this paper is to examine the interactions of hydrology and vegetation dynamics as well as to highlight the role of catastrophe or disturbance in initiating vegetation change in wetlands.

FRESH WATER WETLANDS

In the Northeast many types of emergent and forested wetlands occur on post-glacial sites. Many are dissected by stream courses and underlain by relatively shallow muck type soils. Although primarily of glacial origin, some have been created by human activities. For example, at the Connecticut Arboretum at Connecticut College a four-acre pond 3-4 ft in depth was created over a half century ago in a former post-agricultural semi-open forested seepage-fed depression. Since that time, the pond has progressed from open water to aquatic beds and an emergent wetland. Serving as a setting for the Arboretum's Outdoor Theatre, this pond has traditionally been considered a desirable aesthetic feature. However, since the 1970s the white water lily (Nymphaea odorata), and water shield (Brasenia schreberi) and certain emergents such as pickerel weed (Pontederia cordata) and bur-reed (Sparganium chlorocarpum) have gradually reduced the area of open water. By the early 1980s over two-thirds of the water was covered with water lilies and emergents. In order to arrest this process a draw down was initiated in the early 1980s which continued through the winter with the objective of freezing out the aquatics. In the well-drained areas of the pond the density of water-lily rhizomes was greatly reduced and the following year the pond was relatively free of aquatics except where water shield occurred since it was unaffected. The following year, however, the water lily density increased and water shield became even more abundant. To arrest this natural eutrophication process and maintain a pond, a foot or more of the organic sediments would have to be removed.

Therefore in certain wetland situations there is some degree of predictability in a sequence of processes. Here the trend appears to be toward a return to a semi-open forested wetland somewhat similar to that which occupied the site originally. However, will this site proceed to an upland oak-dominated forest? As a post-glacial depression, the undisturbed vegetation south of the pond is currently a red maple (Acer rubrum)

forested wetland. Theoretically it has had 10,000 years to develop upland vegetation but is still a wetland. In fact, Nichols (1915) in his classic vegetation studies of Connecticut states that it is "doubtful whether the substratum is ever raised sufficiently to produce a truly mesophytic habitat through the operation of biotic factors alone." There may be a tendency in some areas for hemlock (Tsuga canadensis) and yellow birch (Betula lutea)--two species with broad ecological amplitudes and frequencies on upland sites--to also occur in forested wetlands. But replacement by the regional oak forest is unlikely since hydrological and soil conditions are unsuitable to the autoecological requirements of these upland species. Yet some current college biology and environmental science texts still promote this dogmatic traditional concept of wetlands becoming uplands. Since these authors depend upon general ecology texts for much of their information the fault lies in part with ecologists who have failed to correctly interpret vegetation change in wetland systems.

In the Midwest, the prairie pothole emergent wetlands provide another example of the dramatic role of hydrology. Here, disturbance takes the form of severe droughts or muskrat "eat-outs" which are important factors in controlling the dynamics. In these wetlands vegetation zonation is often a distinctive feature. However, just at the word implies, it often represents a relatively stable series of zones or belts and not necessarily a replacement or succession of one vegetation type by another as has so frequently been implied by traditional succession. Each set of species is responding to certain hydrologic conditions and occurs in different micro-sites depending upon the differing autecological requirements of the particular species involved. Thus, under a given hydrologic regime a zonation pattern can persist for a considerable period of time. As is well documented, the prairie pothole country emergent wetlands literally come and go over a five to thirty year period (van der Valk and Davis, 1978). The four-phased cycle in the process includes 1) dry marsh due to severe droughty conditions, 2) regenerating marsh with the return to normal precipitation 3) degenerating marsh due to a combination of high water, disease, insects and senescence of the vegetation, and finally 4) lake marsh in which the area loses most of its emergents partly due to muskrat populations. With the return of another droughty period the cycle is reinitiated since the seed bank is in place to continue the process. There is paleontological evidence that certain of these prairie marshes have persisted for 10,000 years (Weller, 1981).

An analysis of bog dynamics also brings into focus the limitations of traditional succession and climax concepts. Heinselman (1970) in studying the Lake Agassiz peatland in Minnesota, found that the term contributes little to our understanding of the changes occurring in our northern peatlands. He further states "The record of vegetational changes and peatland evolution in the Myrtle Lake peatland cannot be explained by undirectional processes. There has been no consistent trend toward mesophytism, terrestrialization, or even uniformity. Rather there has been a general "swamping" of the landscape, rise of water tables, deterioration of tree growth and diversification of landscape types. Myrtle Lake itself has not been extinguished by 11,000 years of basin filling. The trend is toward landscaping diversity."

A look at smaller peatlands. Post-glacial bogs that have developed in deep isolated depressions or kettle lakes, can further add to the unpredictable aspect of wetland dynamics. Several long-term studies are especially relevant. At Cedar Creek Bog, Buell et al (1968) followed the actual changes across the bog mat to open water over 33 years. They found that the width of the floating mat did not change over this period; however, the width of the various vegetation types did change. The larch-shrub zone expanded outward into the floating sedgemat, greatly reducing its width. An earlier drier period apparently favored the sedge mat development and as the water level rose over the 33 years of observation larch (Larix laricina) trees and shrub cover expanded outward. Yet between 1937-1948 a 10-year period during these three decades of observation the mat increased 1 m suggesting that short term observations can be misleading (Lindeman, 1941).

At Bryants Bog, Michigan, which has been studied since 1917, the mat has advanced into the bog pool in an irregular manner at an average rate of 2.1 cm per year (Schwintzer & Williams, 1974). In 1972 the open water was 76% of its extent in 1926. However, the vegetation did not change in an orderly pattern but rather in response to unpredictable hydrologic change. The advancing leatherleaf (Chamaedaphne calyculata) belt dominant in 1917 was succeeded by a high bog shrub in the drier years and eventually by a bog forest of black spruce (Picea mariana) by the late 1960s. Then in the early 1970s the spruce was killed as the water level rose and leatherleaf was reestablished. In these two bogs the advancing mats of sedges vs. leatherleaf apparently responded differently to fluctuations in water levels as did the advance of woody growth. This only indicates how difficult it is to predict vegetation changes in wetland systems.

At Beckley Bog, Norfolk, Connecticut, a grounded red maple bog forest contiguous to the upland was killed by prolonged flooding as a result of beaver activities. However, the more open sphagnum-leatherleaf floating mat with scattered spruce and larch nearer the open water was relatively unaffected. Historically, beaver

probably played a major role in maintaining many such wetlands. Periodic rises in the water level would serve to retrogress the wetland vegetation development and initiate new processes of change. At Beckley Bog, Van Deusen and Egler (personal communication 1977) found trees in five belts (1) outer marginal red maple, (2) white pine, (3) large spruce, (4) intermediate spruce and larch, and (5) dwarf spruce closest to the open water--all with about 90 rings. Egler asks, "Is that not what you would expect for an area that was totally deforested up to about a century ago?" Past land use history is also an important component in interpreting wetland vegetation dynamics.

In northeastern Connecticut I have observed the loss of the most mature wetland spruce forest in the State (Nichols, 1915) due to beaver or blockage of drainage. However, the bog was not destroyed since the extensive floating mat of vegetation was unaffected by the rise in water level. Both autogenic and allogenic factors are involved in bog development. In the filling process the glacial sands and silts are replaced by lake sediments primarily (gyttja) which are replaced by fibrous or woody peat as the wetland development changes over time. Although autogenic processes may be significant in the long run, allogenic factors such as hydrology, beavers and past land use are also constantly interacting in an unpredictable manner.

On the southeastern coastal plain another typical boggy wetland, the pocosin, illustrates the interaction of hydrology and fire. The vegetation pattern of these distinctive evergreen scrub-pc 1 pine communities is related to the height of water table and fire frequency (Richardson, 1981, 1983). The pocosin vegetation must dry out sufficiently to permit occasional wildfires to sweep through these communities or pond pine (Pinus serotina) with its serotinous cones, would not be a component of the vegetation pattern.

In the Florida Everglades human manipulation of the water regime has had many serious ramifications on the vegetation and associated biota (Niering, 1985). Draining and lowering of the water table have shifted the fire regime from a naturally constructive to a highly destructive factor (Egler, 1952). When the Glades burned prior to man's disturbance, the fires swept through the saw grass when the soil was moist or actually flooded. With a lowering of the water table ground fires which actually destroy the wetland community have become more frequent. Currently nutrient-enriched waters from surrounding agricultural lands are also converting some saw grass areas to cattails.

In the Glades the reproductive success of the wood stork and alligator are also directly related to water levels. Unnatural excessive flooding can inundate the nests of alligator and the absence of dry periods to concentrate fish for the feeding of young wood stork results in nesting failure. In contrast, the Everglades kite prefers high water levels. Thus an extensive matrix of wetland vegetation with different water regimes is needed to provide the ecological requirements of such a diverse wetland fauna.

Cypress swamps represent one further example of the critical need for oscillating water levels. Cypress reproduction is dependent on periodic dry periods when the soil substrate is exposed for seedling establishment (Mattoon, 1916). During the extreme droughty period in 1986 this important event occurred at the Francis Beidler Forest in Four Holes Swamp, South Carolina. Cypress knees can also serve as micro-sites for seedling establishment.

COASTAL WETLANDS

In recent years, rocky shore environments have provided fascinating sites to explore the interaction of near shore marine invertebrate populations and the mechanisms involved in their patch dynamics (Paine and Levin, 1981; Lubchenco & Menge, 1978; Sousa, 1979). Here too, disturbance is a major factor in initiating change and often those populations which became initially established tend to persist as relatively stable populations. The concept of initial faunistic or floristic composition (Egler, 1954) appears highly relevant rather than the traditional relay dynamics frequently associated with succession.

In the Northeast, tidal wetlands offer still another example of the interaction between hydrology and vegetation change. Peat cores document a vegetation development from the intertidal salt water cordgrass (Spartina alterniflora) to salt meadow cordgrass (Spartina patens) with coastal submergence. This appears to be an autogenic process in the traditional succession sense. Along the upland marsh interface there is also a tendency for salt marsh species to replace upland species as the marsh advances landward with sea level rise. However, once the high marsh has developed, oscillations in the vegetation pattern are primarily hydrologically-induced but also involve a complex interaction of factors. Waterlogged, poorly drained sites on the high marsh favor the short term of S. alterniflora. Adverse site conditions can also be induced by mosquito ditching, with the levees or micro-relief changes along the ditches preventing the periodically flooded high marsh from draining. A myriad of vegetation changes can occur once the high marsh has developed, as documented by hundreds of peat cores taken by Orson et al. (1987) in tracing the ontogeny of the Pataguanset marshes in Connecticut, and others taken by the author along the entire Connecticut coastline. A single meter length core, representing 500-1000

years, may show six or more vegetation changes based on preserved rhizomes (Niering et al, 1977). There appears to be no predictable unidirectional pattern. Hydroperiod, changes in micro-relief, accretion rates, salinity, redox potential, sulfide accumulations and other factors make tidal wetland systems too complex to be orderly or predictable (Niering & Warren, 1980). As stated by Miller and Egler (1950), "The present mosaic may be thought of as a momentary expression different in the past, destined to be different in the future, and yet as typical as would be a photograph of moving clouds."

In fact, we are observing these changes today with our long-term studies on the Mamacoke Island Natural Area tidal marsh in the Connecticut Arboretum at Connecticut College. Over the three decades that we have studied this system two major S. alterniflora panne die-outs have occurred, one in the 1960s, and another in 1986-87. The exact cause of such die outs is unknown, however; low oxygen availability and high sulfide levels typical of such sites may be contributing factors. Yet following both events S. alterniflora returned as part of this high marsh panne community.

Another drastic change in a tidal wetland system has occurred at the Wequetequock-Pawatuck tidal marsh (Barn Island Wildlife Management Area) where a saline valley marsh was cut off from full tidal flushing by a dike forming a relatively fresh water impoundment in the 1940s. The Spartina-dominated vegetation was converted to a cattail-Phragmites community. However, within the last decade restoration efforts were initiated to restore tidal flushing. By 1982 two 6-ft culverts were in place and today much of the narrow-leaved cattail (Typha angustifolia) has been replaced by S. alterniflora.

Southward along the Atlantic coast tidal marshes are replaced by mangrove swamps noted for their distinctive belting pattern. In southern Florida, three mangroves--the red (Rhizophora mangle) (most oceanward), black (Avicennia germinans), and white (Laguncularia racemosa) (most landward)--frequently form a belting pattern which has been interpreted as a succession oceanward (Davis, 1940). Egler (1948) questioned this interpretation and, more recently, Ball (1980) found that interspecific competition was an important factor in controlling the zonation. In Panama, Rabinowitz (1975) found that reciprocal transplants can grow well in either zone. She also found that a primary mechanism controlling zonation is tidal sorting of propagules due to their size, rather than habitat adaptation. It appears that, once a given zone reaches equilibrium, it is unlikely to change unless disturbance occurs (Odum et al, 1982).

CONCLUSIONS

Since wetlands are pulsed systems where changing hydrologic conditions can result in concomitant changes in the vegetation and where catastrophic or unpredictable events are frequently occurring, a more open-minded view is required for those concerned with vegetation dynamics. Wetland ecologists are finding traditional succession and climax concepts of limited usefulness and caution all those concerned with wetland change to critically evaluate what is actually occurring in any given wetland based on careful field observations rather than any preconceived ecological concepts concerning how a given system should change. Depending upon the hydrology, wetland vegetation can remain remarkably stable with minimum change for extended periods of time, or, on the other hand, it can be in a constant state of flux.

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The Influence of Hydrologic Maxima and Minima on Wildlife Habitat and Production Values of Wetlands

Milton W. Weller
Department of Wildlife and Fisheries Sciences
Texas A & M University

INTRODUCTION

Variation in rainfall and resulting surface and ground water flows characteristically result in widely varying water conditions in wetlands. In fact, wetlands typically can be differentiated from lakes and other more permanent water bodies by the degree of such fluctuations. These extremes may directly influence wildlife presence and production, and still more commonly influence their habitat values through modification of the plant substrate, food abundance and variety, and physical elements that modify spatial relationships. Common parameters of change are water depth, which also may effect size, and hydroperiod. However, rate of change also is vitally important and may be the primary cause of impact on some species.

Impacts of maxima and minima may be felt at all scales: the organism (or component thereof), population, species, assemblage of species (community), the wetland unit, a complex of wetlands or the landscape level. Impacts at the landscape level are likely to affect all "lower" levels, but individual or even population effects may be common but imperceptible at our usual scale of observation.

The effects of water typically are felt in changes in processes, both biological and physical, and are sometimes observable as changes in rates rather than in all-or-nothing responses. Examples are germination or embryonic development, reproduction, growth, physiological and physical response to drought, water pressure, turbidity or movement, population survival or dispersal, and community characteristics such as species diversity and succession. Other physical responses are especially evident in coastal tidal regimes, or in wind-driven wave action, which may vacillate widely by season and year.

HABITAT PATTERNS TO WHICH VERTEBRATES MUST ADAPT OR RESPOND

Habitat is the place that provides water, food or nutrients, cover, and spatial or social requirements for an organism--plant or animal. To a large degree, its life history strategy evolved with and is still dictated by periodicity in environmental resources such as food or water, and the concept can be readily applied to other influential events such as water fluctuations. These have been classified by periodicity as predictable, unpredictable, or ephemeral (Southwood, 1977) (Fig. 1). Predictable events are those that occur in a regular pattern, whether stable (relatively unchanging) or unstable (fluctuating). Unpredictable variations are those that are erratic in time and degree. Ephemeral fluctuations are those patterns such as spring rains, which are regular in some places, and commonly fluctuate within certain expected values. These events influence habitat quantity and quality, and dictate patterns of mobility and the strategies of habitat use.

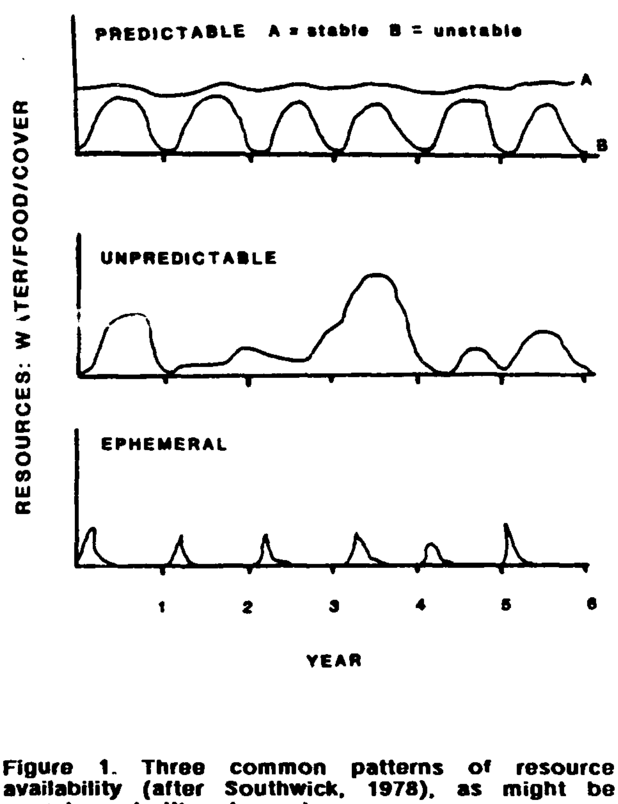

Figure 1. Three common patterns of resource availability (after Southwick, 1978), as might be experienced with water regimes.

Habitat can be viewed as an integration of the spatial component, area or quantity, and the essential resources of food, water, and cover found therein. If resources are lacking, no amount of area is adequate to attract the species for more than a testing period, and this must be considered non-habitat (Fig. 2). With modest resources, some individuals of the species may be attracted into what has been termed suboptimal habitat (Svardson, 1949), and is where one expects to find fluctuating populations and low or erratic production. Optimal habitat is that which is both attractive to the species and results in high levels of production (reproductive fitness) and the resultant maintenance of the population. In some cases--regardless of the size above the minimal--such habitat is so attractive that mechanisms of population control are evident with density dependent reproduction limiting or at least reducing overpopulation. In the case of some herbivores, habitat destruction results (e.g., muskrat eat-out) and habitat quality may be impacted for many years (Errington, Siglin, and Clark, 1963).

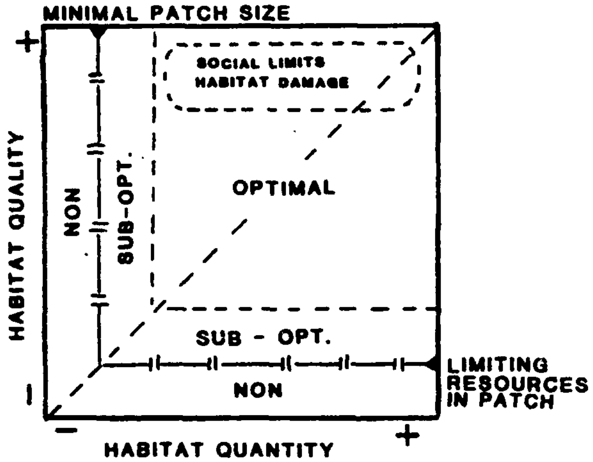

Figure 2. A model of habitat quality and quantity based on minimal and optimal conditions. Exceedingly low quantity (small patch size) or quality (limiting resources) is not attractive as a habitat. Optimal may induce internal population regulatory mechanisms.

Wetland vertebrates select habitat on the basis of their evolved responses and morphological adaptations. In reference to hydrologic parameters, there is some optimal habitat which reflects a balance between water area size, water depth, and hydroperiod as indicated in the model in Fig. 3, but rate of fluctuation (not figured for reasons of complexity) can also be important. Various species differ in their optima, however, as shown by highly aquatic species like wintering or migrant grebes that use open and often deep water, more shallow-water species like blue-winged teal, mammals like muskrats that require a balance between water for protection and emergent plants for food, and by different stages of the life cycle, as for nesting grebes (Fig. 3). Similar responses can be made for plants such as smartweed and cattail.

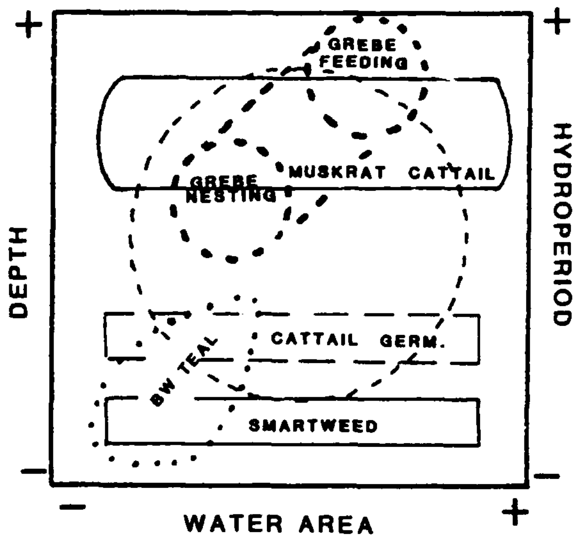

Figure 3. Habitat selection as a function of water area, depth, and hydroperiod. Both plants and animals express preferences and adaptations that determine where and when they are found.

SOME EXAMPLES OF RESPONSES BY WETLAND VERTEBRATES TO HYDROLOGIC MAXIMA AND MINIMA

Low water levels produced by severe drought have been shown to expose muskrats to predation by foxes during food searching (Errington & Scott, 1945) and fish and amphibians to predation by herons and other wading birds (Browder, 1978; Kushlan, 1987). Minimal water conditions also have been known to cause abandonment of nests by birds like coots and ducks that normally nest overwater for protection (Wolf, 1955). The result of such reduced habitat quality and stresses is markedly reduced production, as evidenced in year-to-year fluctuations in populations (Weller and Fredrickson, 1974) and in annual production (Rogers, 1959; Kushlan, 1987). Effects at the community level are evidence of reduced species diversity (richness) at low water periods (Weller, 1981) (Fig. 4), and evidence of interspecific competition when wetland numbers are markedly reduced (Woodin, 1987) (Fig. 5). At the landscape level, the amount of wetland influences populations (Rogers, 1959; Sugden, 1978) so that

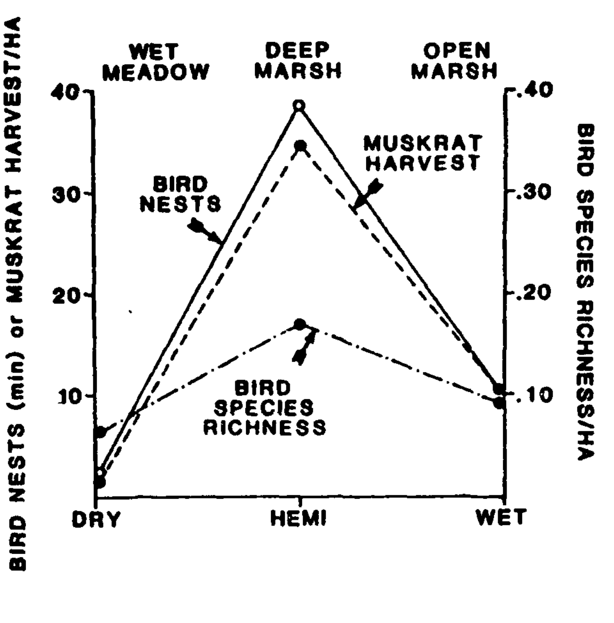

Figure 4. The influence of maximal (flood) and minimal (dry) conditions on the number of nesting birds, bird-species richness, and muskrat numbers on an Iowa marsh (after Weller, 1981).

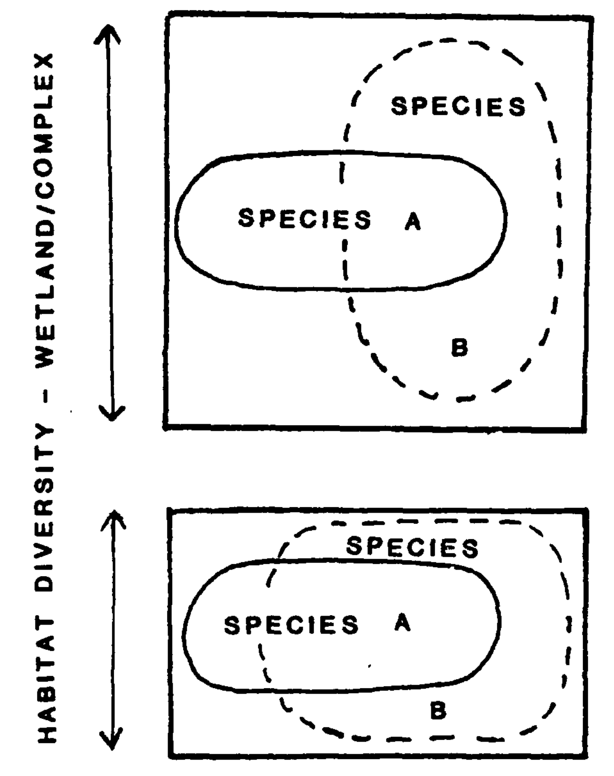

Figure 5. Competition (overlap in resource use) between vertebrates under maximal or optimal (upper figure) and minimal or stressed (lower) conditions as might be experienced with changing water regimes within a wetland or among a group of wetlands (after Woodin, 1987).

populations are minimal during water maxima, which dictates the numbers of flooded ponds, and may cause regional shifts (Derksen and Eldridge, 1980).

Maximal water levels may drown young muskrats (Errington, 1939) or alligators (Kushlan, 1987) in the nest, flood out nests of waterbirds (Harris and Marshall, 1957; Weller and Spatcher, 1965), or expose nests to predation (Rogers, 1959). At the wetland community level, species diversity may be nearly as low as at drought periods, but is made up of more aquatic species (Weller, 1981). At a landscape level, a wetland once productive at normal and minimal water levels (a Utah marsh studied by Weller, Wingfield, and Low, 1958) has been completely inundated and is "non-habitat" for many species under present flood stage of the Great Salt lake (Kadlec, 1984). Wildlife of extensive floodplain forests have been studied with regard to major river impoundments and drainage, with the general effect that major species changes have occurred over time, often due to indirect effects of plant community change (Klimas et al, 1981).

EXAMPLES OF PLANTS AND PLANT COMMUNITIES AFFECTED BY MAXIMA AND MINIMA

Plants are subject to many of the same stresses of water maxima and minima as are animals, but they are more likely to have graded impacts depending on depth, turbidity, temperature and other physical influences. Variation among species is also dependent upon evolved tolerances, ranging from the near-terrestrial to the aquatic. The model shown in Fig. 6 considers selected processes and products of some wetland plants as affected by maxima and minima. Numerous plant processes are affected: Germination is induced by nearly dry soil conditions in shoreline species such as smartweeds (Polygonum spp.) and millets (Echinicholoa spp.) but not in deep water species. Growth of seedlings may be enhanced by minimal water (cattail) and reduced by flooding; mortality of seedlings is common when completely flooded, especially with turbid water or silt deposition (Hosner, 1958). Seed production of shoreline species may be eliminated by drought or flood, although data are sparse on most species. Tuber production in cattail may be enhanced by shallow water or decreased by maximal levels, thereby affecting the plant population and the potential for dispersal. Observations of this type were documented in experimental situations by Harris and Marshall (1963) and are shown in the simple but effective model of germination in relation to season and rate of drying (Fig. 7). Much additional work of this type needs to be done on such life-history strategies to better define patterns, and to enhance management capability as well as

community understanding (Sharitz and Lee, 1985).

	MINIMA				MAXIMA
	DRY (edge)	LOW (shallows)	MEAN (interm.)	HIGH (center)	FLOOD (all)
Germination	TM-	MSA	SA	A	
Seedling growth	TM-	MS	SA	A	
Seedling survival	TM-	MS	SA	A	A-
Plant growth	T	MS	S	A	A-
Seed production	T	MS	S	A	
Tuber production	T	MS	SA	A	A-
Foliage production	T	M	S	A	
Plant spec. richness	T	M	SA	A	
Dominance	T	M	S	A	A-

T - Near-Terrestrial (moisture-loving graminoids)
M - Moist Soil (Sagittaria, smartweeds)
S - Semiaquatic (water-tolerant emergents)
A - Aquatic (submergents, several emergents)
+ - Enhanced
- - Reduced

Figure 6. Some responses to hydrologic maxima and minima by plants with adaptations for various habitats (terrestrial, shallow, aquatic, etc.)

The community patterns measured by species richness are certainly changed by long-term maxima, with a kill of even water-tolerant trees as well as herbaceous vegetation. In some cases, replacement is by less diversity, particularly at the higher water levels. Commonly greater diversity results at low water levels (Weller and Fredrickson, 1974). Little data are available to infer whether major, long-lasting hydroperiods impact dominant vegetation at the landscape level, but my personal observations in field situations and from aerial photographs of prairie wetlands suggests that this occurs on the time scale 3 to 10 years.

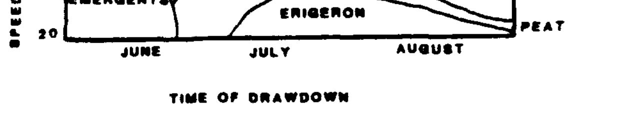

Figure 7. An early but still viable model of seed germination responses to rate and time of drying in relation to soil conditions (Harris and Marshall, 1963).

Recently, we have seen a die-off of mature overcup oak (Quercus lyrata) in east Texas (Fig. 8), a tree that supposedly tolerates continuous flooding for a year or more (Whitlow and Harris, 1970). Because the die-off is of trees immediately surrounding the wetland, and not of higher overcup oaks, flood rather than drought has been identified as the cause. Other observers have recorded similar events with other species, sometimes due to human-induced growing-season flooding (Yeager, 1949; Conner et al, 1981), whereas flooding during the dormant season seems to have less impact at least on trees (Fredrickson, 1982). Many genetic and physiological factors seem involved (Kozlowski, 1984). Others have noted tree mortality due to water shortage (Broadfoot and Toole, 1958). This has led us to a study of lowland plant communities in relation to current water regimes, which provides some interesting relationships that probably are generally true, but rarely does one experience the minima and maxima in such short term studies. It is the unique periodic events that produce mortality, germination, or other responses, and we need to establish long-term monitoring of typical sites to ensure recording of these catastrophic events that probably can change whole communities over landscape level areas.

USING MAXIMA AND MINIMA IN WETLAND MANAGEMENT

Because of the above responses by selected species to hydrologic events, such minima and maxima have been used in the control of nuisance wildlife and fish as well as weeds where water control structures are available. Carp and other "rough" fish that create turbidity and disrupt submergent and emergent plants in wetlands (Threinen and Helm, 1954) are controlled by drainage as the most inexpensive way to enhance a wetland. The use of poisons such as rotenone concurrently assures a more effective kill, but adds enormously to the cost. Extreme water levels also limit population though inhibition of reproduction or increase in predation, but most managers attempt instead to manage by harvest when pelt and carcass values make this feasible.

Plant control by means of flooding is better known, and may be even more effective in conjunction with cutting and flooding (Weller, 1975). Certain herbaceous plants are easily killed this way, whereas willows and other water-tolerant plants may be very hardy, even under severe stress (Weller and Spatcher, 1965, Fredrickson and Taylor, 1982). Flooding when plants are frozen in ice is difficult to arrange but is devastating to dense cattail and other species that become undesirable dominants.

Plant community change for various wildlife

Figure 8. Die-off of normally water-tolerant overcup oaks due to extended flooding (Engeling Wildlife Management Area, TX).

and other functions has been common in wetlands, but is a complex field beyond the scope of this paper. In the same vein, a word of caution is in order lest we forget that any single purpose goal cannot be accomplished in isolation from other species, and that non-target species also will suffer the stress of artificially induced maxima or minima.

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Peat Stratigraphy Evidence of the Influence of Hydrology on Succession in a Freshwater Wetland, Sandwich, Massachusetts

Colen R. Peters
Timson, Schepps & Peters, Inc.

INTRODUCTION

An unusual outcrop exposure and an abundance of fundamental scientific baseline information provide valuable insights into the origin and successional development of a freshwater wetland in Sandwich, Massachusetts. The wetland, located in a gravel quarry, was, at one time, cleared and modified for the growth of cranberries, but this use was abandoned. Subsequent excavation of a gravel pit has exposed a 10-20 vertical foot outcrop of the material underlying the wetland over a distance of several hundred feet. In addition to information available by direct observation, baseline information for the site and immediate area include: a map of the surficial geologic deposits, a geologic interpretation for the origin of the basin, a water table contour map, and radiocarbon dated pollen profiles of nearby peat deposits. Study of this site has provided further information regarding the origin and development of freshwater wetlands in glaciated regions and can be used as a design model for the creation and/or restoration of wetlands in this area.

BASIN CHARACTERISTICS

The freshwater wetland is located two miles south of Sandwich village atop the Sandwich Moraine on Cape Cod, Massachusetts. The wetland was cleared and ditched around the perimeter and across the interior for production of cranberries. Vegetation on the surface of the wetland reflects this drainage: pitch pine (Pinus rigida), red maple saplings (Acer rubrum), highbush blueberry (Vaccinium corymbosum), wool-grass (Scirpus cyperinus), soft rush (Juncus effusus), dewberry (Rhubus hispidus) and, of course, cranberry (Vaccinium macrocarpon).

Recent (1978) aerial photographs indicate the wetland formerly covered approximately 1.8 acres and had a maximum length of approximately 420 ft in the east-west direction and a maximum width of 280 ft in a north-south direction. However, excavation at an adjacent gravel pit encroached into the southern half of the wetland so that the maximum east-west length and north-south width are now approximately 370 and 100 ft respectively.

The wetland was formerly enclosed by the 180 ft contour (NGVD datum). However, this closed depression has been breached for a depth and width of approximately 20 and 50 ft respectively by an excavation of the gravel pit. This was undoubtedly done to drain the wetland and facilitate excavation of the underlying peat as a 10 to 15 ft high outcrop of peat is exposed for a distance of several hundred feet through the center of the wetland.

Water table contour maps (CCPEDC, 1982) indicate the regional water table beneath the area occurs at an elevation of approximately 55 ft (NGVD) or about 125 ft below the surface of the wetland. Consequently the wetland was perched above the regional water table and was dependent on the surface water recharge provided by a 4.6 acre watershed. It is unlikely the entire watershed was an effective recharge area for the wetland given the permeability of adjacent upland soils. The watershed for the wetland is now about half its former size due to the gravel pit excavations.

DEGLACIATION AND BASIN ORIGIN

Three northerly retreating lobes of ice from a continental glacier were responsible for the unconsolidated clay to gravel-size material making up Cape Cod (Oldale, 1982). Radiocarbon dates indicate the central or Cape Cod bay lobe retreated from Martha's Vineyard sometime after 15,300-+ 800 years BP (Kaye, 1964) and was to the north of Boston as early as 14,250-+ 250 years BP (Larson, 1982) or as late as 12,800 years BP (Schafer, 1979). During this 1,000 to 2,500 year period, the Cape Cod Bay lobe occupied present day Cape Cod Bay. From this position the ice lobe fed southerly flowing meltwater streams that deposited sand and gravel on an outwash plain. Isolated blocks of ice left in front (to the south) of the glacier were covered or partially covered by this stratified drift and subsequently melted to form kettles.

The Sandwich Moraine trends east-west across the northern side of Inner Cape Cod. The ridge rises above the outwash plain deposits to the south. It measures approximately 2 miles in width and is characterized by steep hills with heights of 200 to 250 ft (NGVD) and deep depressions that commonly have floor elevations of 50 to 100 ft (NGVD). Oldale and O'Hara (1984) propose and

support with considerable evidence that the moraine formed from readvances of the Cape Cod Bay lobe which thrusted before it (to the south) blocks of the outwash plain deposits or even preglacial strata. These thrust blocks could have been as much as 1.0 km long, 0.5 km wide and 20-30 m thick and may have maintained a distinct sheet-like configuration due to permanently frozen ground or permafrost. Thus the origin of these thrust ridges would have been due to a process similar to a deck of cards sliding over one another. Till overtop on the thrust blocks of stratified drift represents a complete readvance of the glacier over the blocks.

The freshwater wetland which is the topic of this study occupies a basin that occurs between thrust ridges. The wetland basin is enclosed by the 180 ft contour. Other basins at this elevation or with a floor elevation of as low as 100 ft NGVD are numerous. However, few contain freshwater wetlands. Given the occurrence of the regional water table in this area at an elevation of approximately 55 ft NGVD, unique stratigraphic conditions must exist beneath the basins which contain wetlands and thereby provide the necessary hydrologic conditions that permit development of the wetlands.

BASIN STRATIGRAPHY

A variety of criteria have been used for the classification of peats including: geotechnical properties (Landva et al., 1983); degree of plant fiber decomposition such as the von Post system (von Post, 1922) or the U.S. Soil Conservation Service's classification of histosols (Soil Survey Staff, 1975); depositional-water regime (Moore and Bellamy, 1974); microscopic constituents (Cohen and Spackman, 1972) and macroscopic plant constituents (Dachnowski, 1924; Rigg, 1940). The later method of classification, utilizing macroscopic plant constituents, is of great utility in the field, has been widely used in peat resource inventories (Cameron, 1970a, 1970b, 1975; Boothroyd et al., 1979; Peters et al., 1982) and makes use of criteria that reflect the environment of origin, thereby permitting interpretation of successional history (Heinselman, 1963, 1970; Peters, 1981).

Classification based on macroscopic plant constituents is used to describe the peat stratigraphy in the Sandwich Moraine wetland. This system is thoroughly described by Dachnowski (1924). The stratigraphy beneath the wetland was observed at two outcrops in the excavated portion of the wetland. One outcrop measured 11 vertical feet in height and the second measured 13 vertical feet. Peat stratigraphy in both outcrops is very similar, however, the 13 ft outcrop also provided a four foot exposure of the underlying mineral sediment.

The upper foot of the stratigraphic section is comprised of coarse sand and pebbles. Subsequent to clearing of the surface vegetation, this material was emplaced and drainage ditches excavated in it for the production of cranberries.

The coarse sand is underlain by a two foot thick unit of moss peat. The upper one-half foot of this unit has been dewatered by the manipulation of water levels for the cranberry bog and is hardened and exhibits desiccation cracks. The remainder of the unit is comprised of reddish brown, fibric plant remains from Sphagnum moss. Minor amounts of woody material are also present.

A one to two inch thick charcoal unit is present in the moss peat at a depth of 2.5 ft below the surface. The thickness of this unit is very uniform and continuous across the peat outcrop. The thickness suggests that a fire not only burned the surface vegetation in the wetland but also the underlying peat. Underlying this unit down to a depth of 3.7 ft below the surface are four bands of charcoal. Each band measures 0.2 ft thick but contains 3 or 4 distinct layers of charcoal that are less than 0.01 ft thick. These probably represent fires that burned surface vegetation but did not last long enough to ignite the underlying peat.

The lower three bands of charcoal occur at the top of a woody-moss peat unit which extends from 3 to 5 ft below the surface. This unit is also reddish brown in color but is of sapric texture and contains a greater abundance of woody material including twigs and limbs from 1/2 to 1 inch in diameter.

Below the woody-moss peat from a depth of 5 to 8 ft is brown wood peat in a matrix of highly decomposed or sapric material. Limbs 1 to 3 inches in diameter and logs up to a foot in diameter are common.

At the 13 ft high outcrop, wood peat lies directly above one foot of leached, coarse sand. Although lesser in height, the 11 ft outcrop is located nearer the deeper part of the basin. Additional peat units occur beneath the wood peat before the leached coarse sand is encountered.

At the 11 ft outcrop, from 8 to 8.8 ft below the surface, the wood peat overlies a dark brown, sapric reed-sedge peat. Some small twigs occur in this unit but highly decomposed remains of reeds, sedges and/or other emergents predominate. The reed-sedge peat is underlain by 0.2 ft of limnic peat or gyttja. The limnic peat is greasy in texture and contains highly decompose rganic material with silt and clay. The limnic p t in turn overlies one foot of coarse sand. As in the case of the 13 ft outcrop, the remaining 0.9 ft down to a depth of 10 ft below the surface is leached coarse sand. Crumbly subgranular pebbles and fragments of wood

occur in the leached sand at both outcrops.

The leached sand overlies several feet of till. The upper foot of till is hard and dark black in color due to cementation by ferrous oxide and manganese oxide deposits. This grades into till that is more friable and reddish brown in color from mineralization by ferric oxide deposits. The stratigraphy of both outcrops is summarized in Figure 1.

FRESHWATER WETLAND SUCCESSION IN THE BASIN

A radiocarbon date of organic material at the base of the Sandwich Moraine Wetland has not yet been obtained. However, other studies show that within a few hundred years of deposition a glaciated area can be vegetated by tundra plants (Daubenmire, 1968; Wright, 1981). From another site on Cape Cod, Winkler (1985) shows through radiocarbon dated pollen profiles of a 4 meter long core taken in 18.2 m of water that a tundra-spruce parkland plant community had become established by 12,000 BP and had existed for approximately 500 years before being replaced by a boreal forest. This may provide an indication of the early plant community at the Sandwich Moraine site but obviously confirmation with pollen analysis and radiocarbon dating is required.

Based on the existing stratigraphic exposures at the Sandwich Moraine Wetland, it is possible to interpret the successional development of the wetland independent of the time of origin. Basal limnic peat indicates that a shallow body of water initially accumulated in the deeper parts of this till mantled depression. The dense till, having a low permeability, may have caused the surface water draining into the depression to be perched, permitting the development of a freshwater pond plant community. Until radiocarbon dates are obtained, it cannot be determined whether the till was initially impermeable enough to maintain the pond in the depression or whether this ultimately occurred by cementation of the till with ferrous and manganese oxide deposits subsequent to repeated perching and drawdown of water in the depression.

The reed-sedge peat over limnic peat indicates the pond succeeded into a marsh. Probing of the undisturbed parts of the wetland indicates the outcrops exposed by the gravel pit operation do not occur in the deepest part of the basin. Thus both the limnic peat and reed-sedge peat units are likely to be thicker towards the center of the basin. However, the marsh did not necessarily occupy the entire basin as water depths were shallow enough to permit the growth of woody wetland vegetation and thereby the accumulation of wood peat directly on top of mineral soil.

Woody wetland vegetation in the form of a shrub swamp or a wooded swamp, as suggested by 1 ft diameter tree trunks, then occupied the basin to form 3 vertical feet of wood peat. This was followed by a transitional period in which 2 vertical feet of woody moss peat was deposited either by the persistence of a shrub or wooded swamp or with the development of a bog community. Charcoal indicates fires were frequent throughout this period and may have been responsible for the ultimate establishment of a bog which is represented by 2 vertical feet of moss peat. A 0.1 ft thick charcoal unit occurs near the base of the moss peat and probably represents a fire which not only burned the vegetation on the surface of the wetland but also ignited the underlying peat. Due to the stripping of vegetation and placement of fill atop the moss peat unit, it is no longer possible to determine what type of wetland community occurred at the site immediately prior to the creation of the cranberry bog.

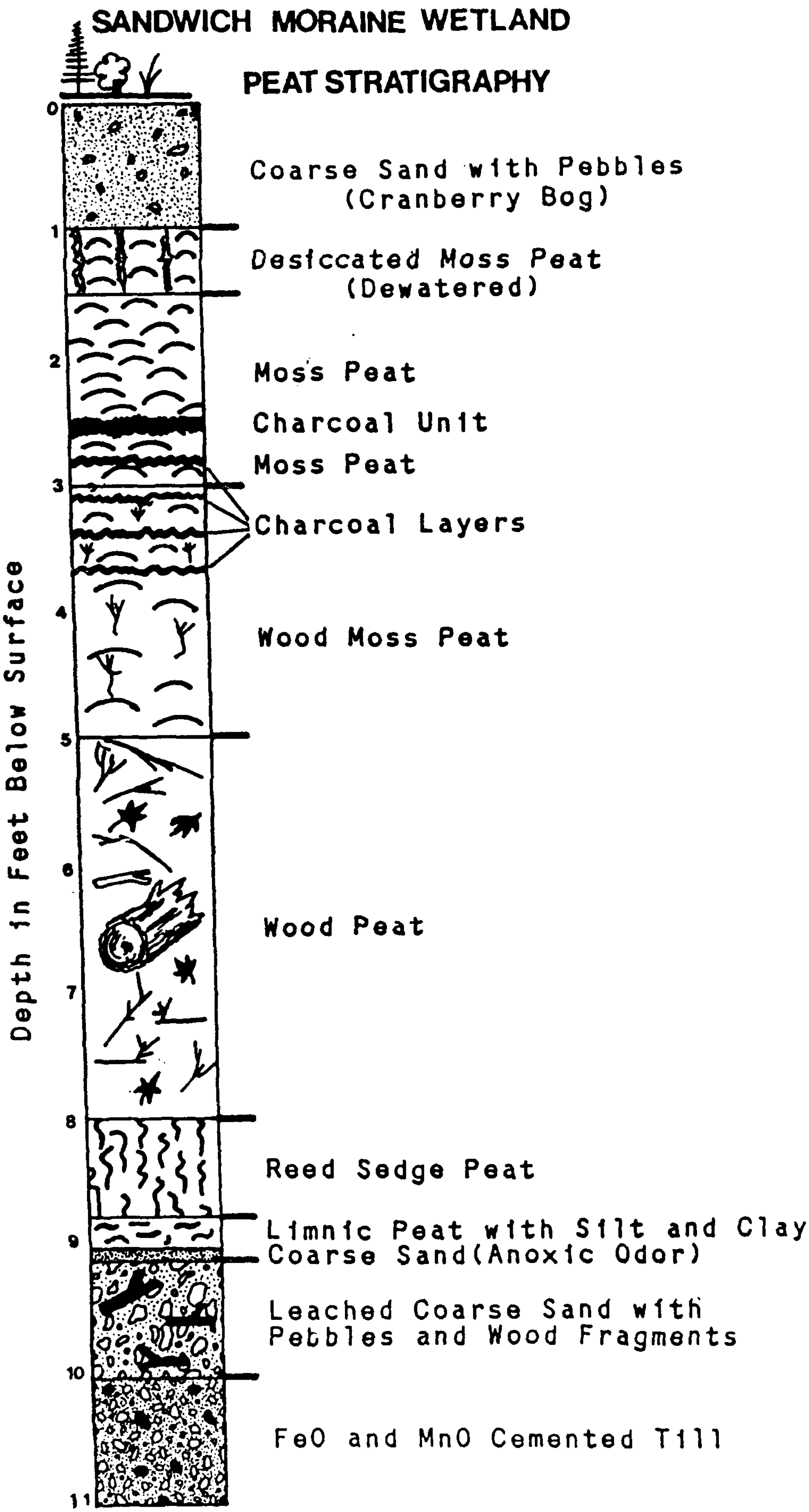
RELATION OF THE WETLAND TO MODELS OF WETLAND DEVELOPMENT

Two models of wetland development are described for glaciated North America: (1) hydroseral succession (Dansereau and Segadas-Vianna, 1952; Daubenmire, 1968; Moore and Bellamy, 1974) and (2) paludification (Dachnowski, 1924; Heinselman, 1963, 1970; Moore and Bellamy, 1974).

An *oligotrophic* (nutrient-poor, oxygen-rich) pond or lake occupying a closed glacial depression exemplifies the initial stage of hydroseral succession (Daubenmire, 1968). Submergent and floating plants become established along the shore and in the areas where terrigenous deposition has reduced water depth to less than 2m. The pond or lake may change from oligotrophic to eutrophic (nutrient-rich, oxygen-depleted) when the remains of submergents and surface plants begin to accumulate as limnic peat. Further filling of the lake/pond permits the centripetal establishment of emergent vegetation represented by reed-sedge peat.

Anchored or floating marginal mats characterize the fourth stage of hydroseral succession. A thick, tangled mat of peat supporting moss and sedges originates along the margins of the wetland and encroaches towards the center. Moss peat accumulates in nutrient-poor water regimes while sedge peat is deposited under more nutrient-rich conditions (Daubenmire, 1968). Shrubs and trees invade whenever and wherever water depth will permit growth.

Paludification refers to "the process of bog [wetland] expansion caused by gradual rising of the water table as peat accumulation impedes drainage" (Heinselman, 1963). Peat stratigraphy


SANDWICH MORAINE WETLAND
PEAT STRATIGRAPHY
Depth in Feet Below Surface
0
1
2
3
4
5
6
7
8
9
10
11
Coarse Sand with Pebbles (Cranberry Bog)
Desiccated Moss Peat (Dewatered)
Moss Peat
Charcoal Unit
Moss Peat
Charcoal Layers
Wood Moss Peat
Wood Peat
Reed Sedge Peat
Limnic Peat with Silt and Clay
Coarse Sand(Anoxic Odor)
Leached Coarse Sand with Pebbles and Wood Fragments
FeO and MnO Cemented Till

resulting from paludification in the Lake Agassiz region of north-central Minnesota consists of basal forest peat overlain by moss peat, indicating forests invaded by bog. Basal sedimentary peat is absent or only locally significant (Heinselman, 1970). Wetlands resulting from paludification in Quebec, Ontario and Northern Europe (Dansereau and Segadas-Vianna, 1952; Moore and Bellamy, 1974) originated as lake-filled basins that expanded over surrounding upland. In the North American and European examples, wetlands progress from a minerotrophic stage (nutrients and water derived from outlying, mineral soil) through more nutrient-poor conditions to an ombrotrophic stage (nutrients and water derived from rain).

At the Cape Cod site, given the sequence and thickness of wood moss peat in both outcrops and the occurrence of wood peat directly over the leached sand in the 13 ft outcrop, both outcrops clearly exhibit evidence of the wetland developing by paludification. Given that the upper 7 to 9 ft of both outcrops is comprised of either wood or moss peat, this process of development appears to have been very significant throughout the history of the wetland. Peat stratigraphy in the lower two feet of the 11 ft outcrop contains a sequence of units that may represent hydroseral succession; however this must be confirmed by determining the extent, thickness, and sequence of these units in the deeper part of the basin.

SUMMARY

On Cape Cod, Massachusetts, within the Sandwich Moraine, the stratigraphy beneath a freshwater wetland is cross-sectionally exposed in two outcrops measuring 11 and 13 vertical feet in height. The bottom of the wetland is perched approximately 100 ft above the regional water table. The most complete stratigraphic section appears in the 11 ft outcrop and from bottom to top includes: iron and manganese oxide cemented till; 1 ft of leached coarse sand with wood fragments; 0.2 ft of limnic peat; 0.8 ft of reed-sedge peat; 3 ft of wood peat; 2 ft of woody moss peat containing numerous charcoal layers; 2 ft of moss peat with a 0.1 ft thick charcoal unit at the base; and at the top 1 ft of coarse sand from a former cranberry bog. The 13 foot outcrop is located nearer the edge of the basin and exposes more of the underlying till which is directly overlain by leached coarse sand and then wood peat. Thickness and sequence (wood peat, woody moss peat with charcoal, moss peat and coarse sand) of the remainder of the outcrop are similar to the 11 ft outcrop. The thickness, sequence and types of peat present show that paludification has had a major role in the development of the perched wetland as it succeeded from a shrub or wooded swamp to a bog. Limnic peat and reed-sedge peat near the center of the basin suggest hydroseral succession may have been important in the early stages of the wetland's development.

Further research is necessary to better understand the successional development of this wetland basin and other wetlands. The water budget of perched wetlands requires further study; especially the relationship of watershed area to wetland area coupled with study of the geology and hydrogeology of basins in similar settings that lack wetlands. Stratigraphic studies beginning at the time of basin origin to the present need to be conducted to document successional development. These studies should be accompanied by radiocarbon dated pollen stratigraphy so that climatic and temporal influences on wetland succession can be assessed. This information is particularly necessary from the base of peat deposits to determine the genesis of the wetland and whether the basin from the onset permits wetland development or whether development of the wetland modifies the basin and thereby maintains development.

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Plant Community Boundaries and Water Levels at Lake Hatchineha, Florida

Michael J. Duever and Jean McCollom
National Audubon Society

Lou Neuman
Florida Department of Natural Resources

INTRODUCTION

In early 1985, the authors of this paper became involved in a study to help define the boundary between public and private land along the shoreline of Lake Hatchineha in central Florida. The study was triggered by the action of a lakeside landowner who began to log a portion of the cypress forest which borders much of the lake. In many parts of the United States with greater relief than Florida, there may be relatively little difference in the location of the public/private boundary drawn by the state and a private landowner along a major water body. However, because of the low topographic gradients found in Florida, these differences of opinion can involve distances perpendicular to a lakeshore of a mile or more of valuable lakefront property.

At Lake Hatchineha, a number of circumstances complicated the definition of this boundary between public and private property, formally called the Ordinary High Water Line. Lake water levels have been artificially regulated since 1964, but the lake levels relevant to ownership of the shoreline are those that occurred prior to that date. Prior to regulation, annual water level fluctuations were normally between 3 and 6 ft and, during periods when droughts or floods occurred, these annual fluctuations could approach 10 ft. Barrier bars are also present along portions of the lake subject to heavy wave action. These long, narrow geologic structures are bordered on all sides by plant communities that were normally inundated for extended periods during most years prior to 1964. Barrier bars are not continuous around the lake or with nearby uplands. Where they occur, there is disagreement as to whether they represent the shoreline of the lake, and therefore the line between public and private property, or whether they are merely islands in the lake.

Due to the potential precedent-setting nature of this case for the determination of shoreline ownership of many other lakes in the state, the Florida Department of Natural Resources decided to apply a variety of approaches in their efforts to determine the Ordinary High Water Line at Lake Hatchineha. This was done by utilizing a specialized team disciplined in hydrology, surveying, geology, biology, forestry, and ecology. Lake water level data for the period of record, 1942-1975, were analyzed. Nine transects were established perpendicular to the lakeshore at various points around the lake. Along each transect ground surface elevations were determined. Trenches were dug through portions of the barrier bars and exposed soil profiles were analyzed to characterize development of the bars. Soils and plant species composition were examined to determine the distribution of wetland communities along the transects. Our contribution to the study, and what we will discuss in this paper, involved the determination of the elevations of boundaries between all major structural plant community types along each transect. Where woody vegetation was present in a community, we also aged selected trees to determine whether the community originated before or after 1964.

The objective of these studies was to identify that portion of the shoreline that was regularly inundated during most years prior to regulation and which would thus be considered part of the lake, as opposed to part of the adjacent uplands. The question then became what constitutes regular inundation. Over the past decade scientists have obtained hydroperiod (annual period of inundation) data for major community types at a number of relatively undisturbed sites in Florida. Our work has shown that hydroperiod is one of the dominant factors controlling the distribution of major freshwater wetland community types in Florida. Thus, we felt that by identifying the major plant community boundaries as they now exist at Lake Hatchineha and verifying which were and were not present before 1964, we could relate preregulation annual periods of inundation to specific elevations along the entire lakeshore. While not resolving the question of what constitutes regular inundation, it at least provides some solid data on what lands would or would not be affected if a decision was based on one or another specified annual period of inundation.

METHODS

Major plant community boundaries were identified on the basis of the location of the "community edge", the line beyond which the species

representing the major structural components of the community are no longer dominant. This "community edge" approach allowed a more accurate assessment of how these species were interacting with their environment than would the use of one or several scattered individuals that could exist outside their normal habitat as a result of rare or unusual circumstances.

Each transect was visited and the position of all major community boundaries along the main transect line were determined. At four sites additional spurs from the main transect were established to include examples of some boundaries which were not available along the main transect. The elevation above mean sea level (NGVD) of each boundary was determined from surveys conducted by Florida Department of Natural Resources staff. Selected trees were cored and approximate ages were determined from annual ring counts.

The average annual period of inundation was then estimated for each community boundary elevation from the 1942-1964 elevation - duration curve developed by J.H. Hartwell (pers. comm.) for Lake Hatchineha.

Complete aerial photographic coverage was obtained for the Lake Hatchineha area for 1941, 1952, 1968 and 1983, and partial coverage for 1958. These photos were examined to evaluate both natural and man-caused changes in the vicinity of each transect for the period from 1941 to 1983.

RESULTS AND DISCUSSION

Transect Plant Communities

The dominant structural types of plant communities associated with Lake Hatchineha, in order of increasing elevation, are open water, deepwater marsh, cypress forest, shallow marsh, pine forest, live oak forest, and palmetto thicket. The more ecologically diverse transects were those which included a well developed barrier bar and extended beyond the barrier bar to an elevation supporting upland vegetation. The open water and deepwater marsh communities were present along all transects.

Transect 1

Transect 1 was the most diverse. It supported the following communities: cypress forest (both along the front and back of a high barrier bar), shallow marsh, pine forest (recently logged and a short distance from the main transect), live oak forest (both on top of the barrier bar and inland beyond the shallow marsh), and palmetto thicket. A short spur also ran to a well developed forest of slash pine, Pinus elliottii Engelm., south of the transect. This stand was in a topographic position essentially identical to the logged pine stands adjacent to the main transect. There was a distinct zone with only a sparse herbaceous ground cover on both the front and back of the barrier bar between where the cypress, Taxodium distichum (L.) Rich., and live oak, Quercus virginiana Mill., were rooted. Apparently conditions in this zone were not suitable for the establishment of either cypress or live oak, but their continuous canopy limited the development of a shallow marsh community which would normally occupy this topographic position. There was also a narrow transition zone between the cypress and shallow marsh which was dominated by smaller scattered cypress and wax myrtle, Myrica cerifera L., intermixed with open areas of marsh.

Transect 2

The main line of Transect 2 was among the least diverse sites studied. It supported only shallow marsh and live oak forest. We established a spur from the main transect that intersected pine, live oak, and palmetto communities.

Transect 3

Transect 3 also included a relatively small number of communities. There was a high barrier bar occupied by live oak and some cypress. Cypress were also found on both sides of the barrier bar, but more densely behind it. On both sides of the barrier bar beyond the cypress communities, the land surface dropped away to deepwater marsh.

Transect 4

Construction of a dike adjacent to the shore of Lake Hatchineha between 1941 and 1952 at the site of Transect 4 created a highly altered situation. Only the deepwater marsh could still be considered an intact part of the system. Since only one undisturbed habitat was present, no useable boundary information was available from this transect.

Transect 5

Transect 5 had only a low barrier bar at the lake edge. The cypress forest which began at the edge of the deepwater marsh was virtually continuous up to the shallow marsh. Most of this forest along the transect had been logged recently. There was a narrow transition zone between the cypress and the shallow marsh that was occupied by a mixture of scattered small cypress and pine, clumps of wax myrtle, and open areas of marsh. Beyond the marsh was a dense forest of small slash pine, and finally a generally open area with thickets of palmetto, Serenoa repens (Bart.) Small, gallberry, Ilex glabra (L.) Gray, and scattered sabal palm, Sabal palmetto (Walt.) Lodd. ex Schultes, live oak, and slash pine.

Both of these last communities contained numerous pine stumps from trees larger than those now present.

A spur was run from this transect southeast to an area supporting a fairly well developed stand of slash pine along the upper margin of the shallow marsh. Transect 5 SPUR (which ran inland from the lake at this point) showed a sequence of plant communities similar to those on Transect 5, except for the absence of the transition zone between the cypress and the shallow marsh.

Transect 6

Transect 6 had a very low barrier bar. A cypress forest, most of which had been logged, extended from the deepwater marsh to the shallow marsh. There was no transition zone at the upper margin of the cypress forest. The upper margin of the shallow marsh abruptly adjoined a dense live oak forest with a very open understory.

Transect 7

Transect 7 had a barrier bar of sufficient height to support one live oak in the vicinity of the main transect. Since we did not feel this one individual constituted a change in community type, we described the cypress forest as being continuous to the bay on the inland side of the barrier bar and then continuous from the far side of the bay to the shallow marsh. The larger cypress trees inland from the barrier bar and the bay had been logged, but not recently, since the present canopy was fairly well closed. The logged trees appeared to be of similar age to those remaining. Transect 7 SPUR began in the bay. The bay plant community was more similar to deepwater marsh than shallow marsh. The cypress forest changed rather abruptly into a broad shallow marsh which in turn changed abruptly into a dense live oak forest with an open understory. As one moved into the live oak forest, clumps of palmetto gradually became a more dominant understory component.

Transect 8

Along the edge of the lake were numerous cypress with up to 3.5 ft of root exposure. The barrier bar at Transect 8 was fairly high. It supported several large live oaks dispersed among a number of cypress and did not represent a distinct community. The cypress forest behind the barrier bar was relatively narrow, and most of the larger trees had been logged. The remaining smaller cypress graded into shallow marsh, which in turn graded into a dense live oak forest with an open understory.

Transect 9

There was a high barrier bar at Transect 9 which supported a well developed live oak forest. While a dense cypress forest existed along the front of the barrier bar, the community behind it was more comparable to the transition zone normally encountered between cypress forest and shallow marsh. Little of what remains behind the barrier bar could be considered undisturbed. Most of the area was originally an extensive shallow marsh. In 1958 ditching was begun and the area had been completely diked and ditched prior to 1968. Regular pumping has apparently maintained abnormally low water levels for a number of years within the area confined by the dike. The average water table immediately outside the dike would also be unusually low due to these drainage activities. Thus, plant communities could be expected to occur at lower than normal elevations in the area behind the barrier bar, particularly if they had become established since 1968.

Plant Community Boundary Elevations

The open water - deepwater marsh boundary was located at elevations of 46.3 - 47.5 ft at the six sites for which we had data (Fig. 1A).

At the six sites without high barrier bars, the deepwater marsh - cypress forest boundary was located at elevations of 49.8 - 50.5 ft (Fig. 1B). It was located at 50.0 - 51.7 ft at five sites with high barrier bars.

Four cypress forest - shallow marsh community boundaries were located at a fairly consistent elevation of 51.1 - 51.7 ft (Fig. 1C). There was one additional boundary value of 52.2 ft on transect 8, and the one distinct cypress - transition zone boundary was at 51.4 ft. The upper limit of the cypress forest varied from 53.4 to 54.1 ft on the lakeside of 3 barrier bars, and from 52.3 to 53.2 ft on the inland side of 2 barrier bars.

The boundary between the shallow marsh and the slash pine forest varied from 53.6 to 55.0 ft at 3 sites (Fig. 1D). At one additional site it was located at an elevation of 52.6 ft. While the trees at this last site were fairly large (10-13 in), they commenced growth at this elevation after water level regulation began in the mid-1960s. All of the trees at the other sites had begun growth prior to regulation and were found at elevations at least 1 foot higher.

The seven shallow marsh - live oak forest boundaries were the most variable, ranging from 54.4 to 56.2 ft (Fig. 1E). The lower boundaries of live oak forests on 3 barrier bars extended from 53.7 to 56.8 ft on the lakeside of the bars and from 53.2 to 55.2 ft on their inland sides.

At the three sites with adjacent live oak forest and palmetto thicket communities, their boundary elevations were 57.2 - 58.8 ft (Fig. 1F).

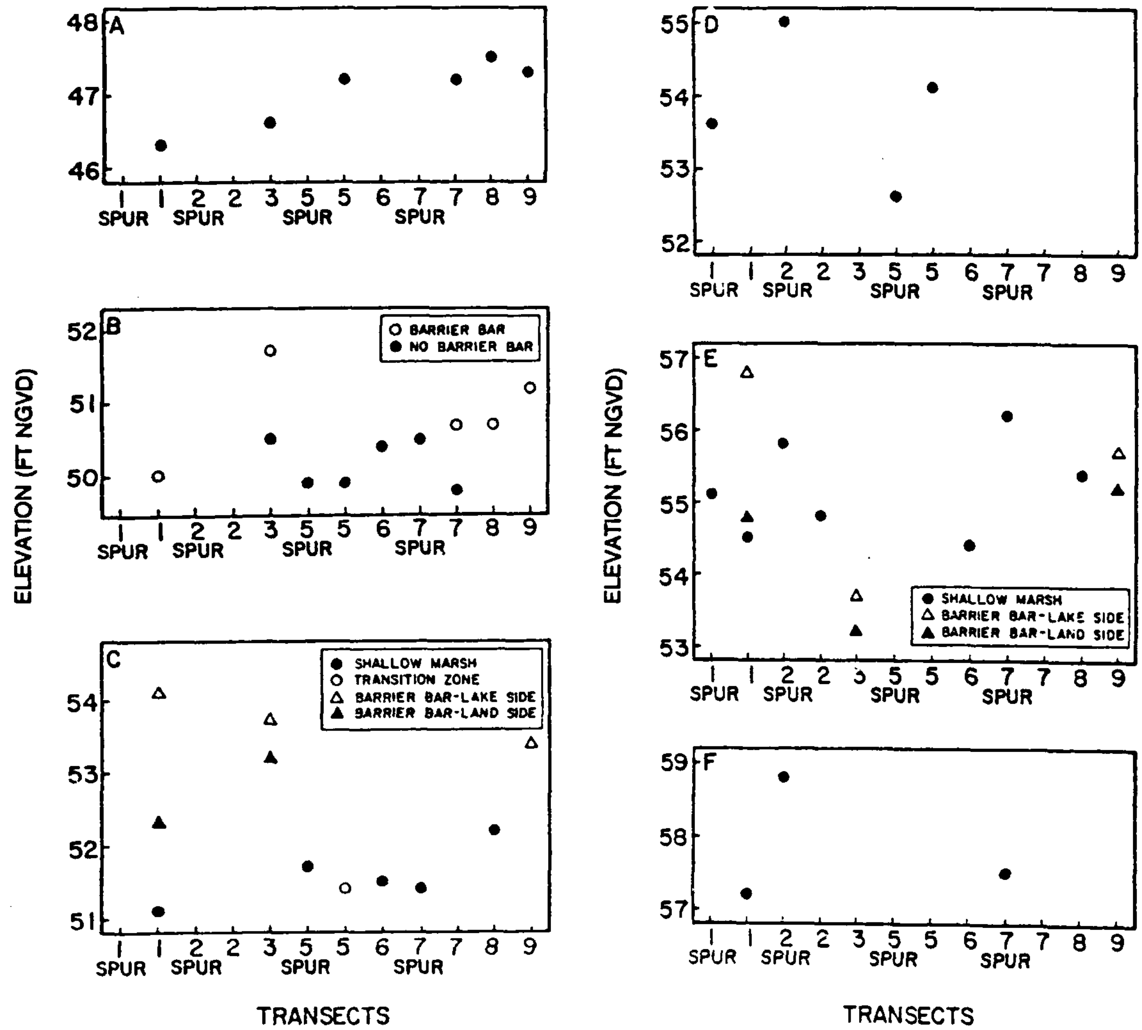

Figure 1. Elevation of community boundaries: A) open water boundary with deepwater marsh. B) deepwater marsh and cypress forest boundary on sites associated and not associated with high barrier bars. C) upper cypress boundary with plant communities or position on the barrier bar. d) upper shallow marsh boundary with slash pine forest. E) lower live oak forest boundary with shallow marsh boundary and position on the high barrier bars. and F) upper live oak forest boundary with palmetto thicket.

Barrier Bar Influences

During our data analysis we found that community boundary elevations associated with the higher barrier bars were quite variable, and were frequently out of line with more consistent patterns observed at locations around the lake where barrier bars were low or absent (Fig. 1B, 1C, and 1E). Boundaries were also consistently higher on the side of the barrier bar facing the lake compared to the same boundaries on the other side of the same barrier bar (Fig. 1C and 1E). In some cases, cypress and live oak which normally occur in very different topographic positions were found intermixed on a barrier bar.

A partial explanation for these seeming discrepancies is that the barrier bars are not stationary, but moving landforms. The processes involved are some mix of water and/or wind action which results in the erosion of the ground surface in one place and its accretion in another. The effects of this movement on the plant community can be seen in exposed roots or buried trunks, which indicate some individuals currently exist at ground surface elevations either much lower or higher than when these individuals first became established. Thus, when one measures the ground surface elevation in relation to the occurrence of such an individual, and then attempts to discuss these data in relation to a species' tolerance of inundation, it is possible to come up with very inappropriate conclusions.

Another factor that could affect the distribution of species on the barrier bar is the direct

effect of waves on vegetation. Heavy wave action on the front of the barrier bar could physically remove individual trees and could maintain higher soil moisture conditions there. Also, cypress seeds that had been in the water for some time and were thus more likely to germinate could be placed higher on the lake side of the barrier bar by high waves. For these reasons, deepwater marsh - cypress and cypress - live oak

Our estimated hydroperiods generally corresponded quite well with those reported for comparable undisturbed habitats in other studies (Duever et al., 1975; Duever et al., 1978; Duever et al., 1986; Gunderson and Loope 1982a, 1982b, 1982c; Gunderson et al., 1982; Pesnell 1977). However, variations in terminology, site history, or community characteristics needed to be accounted for before these similarities were apparent. Thus, there appears to

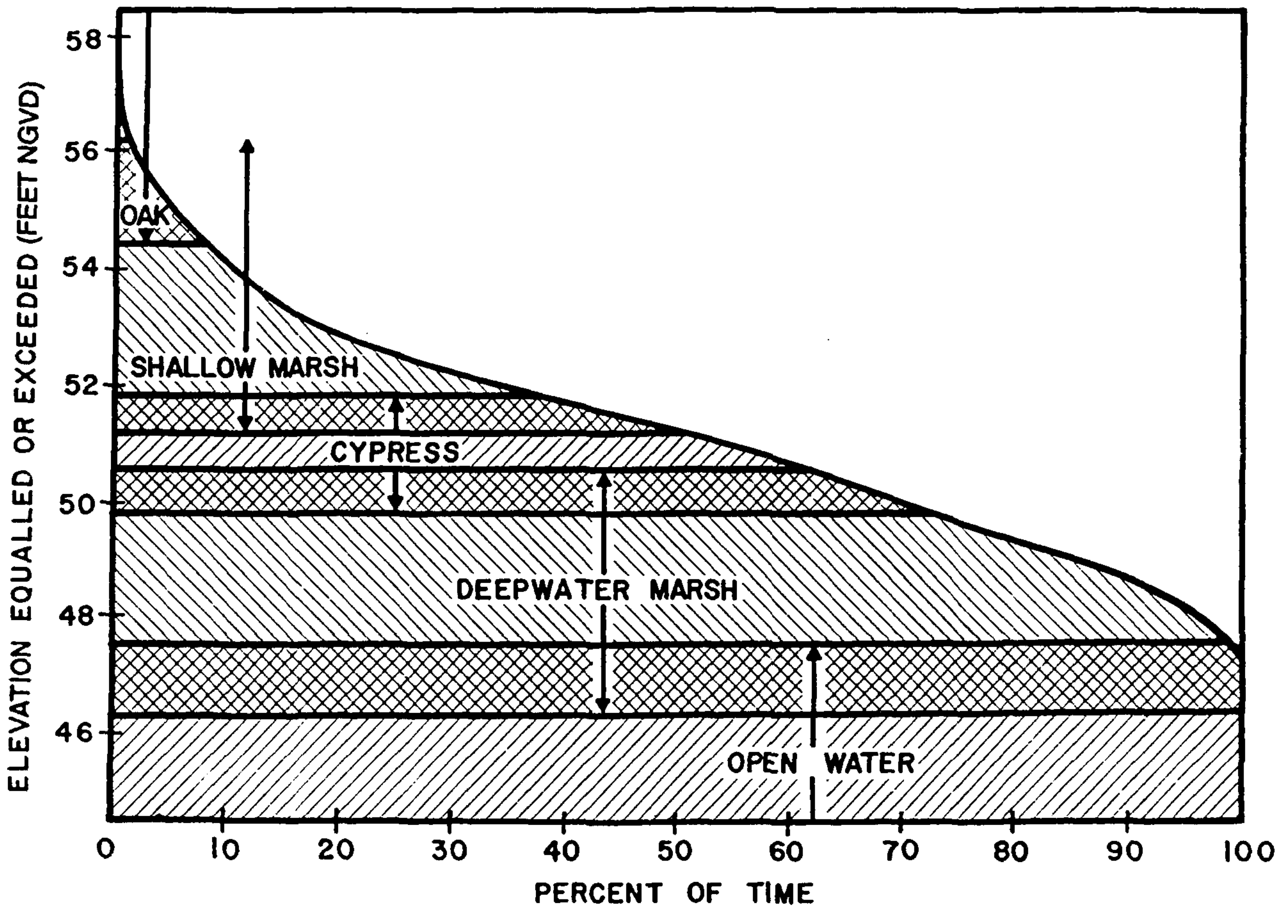

Figure 2. The distribution of major Lake Hatchineha plant communities in relation to J.H. Hartwell's elevation-duration curve, using daily water level values from October 1942 to September 1964. The position of the pine forest - shallow marsh boundary is not shown.

community boundary elevations from sites associated with high barrier bars have been presented for discussion purposes only, and are not included in the following analyses.

be a sound basis for using community boundary elevations at Lake Hatchineha to define environmental boundaries that reflect long-term hydrologic conditions.

Hydroperiod

We estimated the range of hydroperiods for each community boundary. The estimates were made by taking the percent time inundated for each boundary elevation from Hartwell's daily water level elevation-duration curve for the pre-regulation period 1942-1964, and converting it to days per year.

The Ordinary High Water Line

The palmetto boundary at 57.2 ft approximates the point above which water does not normally stand above ground at Lake Hatchineha. Hartwell's 1942-1964 elevation-duration curve identifies an elevation of 57.0 ft as a point above which flooding does not occur at Lake Hatchineha (Fig. 2). If a criterion of zero days inundation during an

average year is selected for the Ordinary High Water Line, then an elevation of approximately 57 ft would best define it. The next lower applicable boundary based on our research would be the lower edge of the live oak community which is inundated for 4 to 29 days/yr. A slightly lower boundary would be that between the shallow marsh and slash pine communities at 53.6 - 55.0 ft and inundated for 18 - 47 days/yr or 5 - 13 percent of the time. Among these last two boundaries, the former would seem the better choice since, based on aerial photography, it has been a stable boundary at Lake Hatchineha for at least the last 40 years.

At the other extreme, if open water without emergent vegetation, which suggests permanent inundation, were the criterion, then the appropriate elevation would lie at an approximate elevation of 47 ft. This results from the deepwater marsh - open water boundary occurring between 46.3 and 47.5 ft., and Hartwell's 1942-1964 elevation-duration curve, which identifies 47.0 ft as the elevation below which permanent flooding occurs at Lake Hatchineha (Fig. 2). Higher community boundaries are the deepwater marsh - cypress boundary which is inundated for 230 - 275 days/yr, or the cypress - shallow marsh boundary at 140 - 185 days/yr.

The determination of the Ordinary High Water Line at Lake Hatchineha ultimately rests with the legal interpretation of applicable laws and relevant information. Hopefully, the results of these soil, hydrology, and vegetation studies will assist the courts in reaching an informed decision.

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Hydrologic Regime of a Floodplain Wetland in Massachusetts

Nancy C. Suurballe
U.S. Geological Survey

INTRODUCTION

Floodplain wetlands play an important role in the hydrology of stream systems. These wetlands influence streamwater quality through complex interactions among biological activity, soil, and aquatic chemistry, sedimentation rates, and hydrology. To understand the structure and function of floodplain wetlands, a quantitative understanding of the hydrologic regime is needed. This paper presents the results of a study to determine a comprehensive water balance through a wetland/stream system and to relate changes in streamwater quality to different hydrological and seasonal conditions. The paper describes the seasonal influence exerted by a wetland under various hydrological conditions and provides a basis for improved predictions of water quality along stream reaches in wetlands. The research that is the basis of this paper was done by the U.S. Geological Survey in cooperation with the Massachusetts Department of Environmental Quality Engineering, Division of Water Pollution Control.

PHYSICAL SETTING

The study area is a palustrine wetland located in the headwaters area of Natty Pond Brook in Central Massachusetts. Nine kilometers of stream channel flow through the wetland, draining a total of 2 square kilometers of wetland and 12 square kilometers of upland. The wetland is located within a drainage basin composed mainly of glacial till. Most of the wetland area is underlain by organic soil (peat), which is, in turn, underlain by stratified glacial drift. The peat layer ranges from 1.2 meters to more than 6.7 meters; its composition ranges from live, undecomposed mosses with a relative high fiber content and low bulk density (at the surface) to relatively well-decomposed herbaceous peat with low fiber content and high bulk density (at greater depths). The stratified drift is composed of sand and gravel deposits ranging from about 10-12 meters in thickness. The wetland vegetation community is composed predominantly of forested and scrub-shrub species. Forested types account for 60 percent of the vegetation, and scrub-shrub account for 30 percent. The remaining 10 percent of the vegetation is represented by emergent, open water, riverine, and mixed wetland vegetation types.

WETLAND WATER BALANCES

Wetland water balances were determined for the period November 1984 through October 1985, and the period November 1985 through October 1986. The equation used to express the water balance of the wetland/stream system is

$$ULI + P = SRO + ET \pm \Delta ST$$

All terms represent water volumes and are defined as:

ULI upland surface and ground water inputs to wetland

P direct precipitation on river and wetland

SRO surface-runoff output at the downstream limit of wetland basin

ET evapotranspiration loss by wetland

ΔST changes in surface and ground water stored in wetland

The components in the water balance equation represent direct inputs and outputs to the river wetland system. Precipitation inputs (P) were calculated based on National Oceanic and Atmospheric Administration data from a rain gage located within one mile of and at a similar elevation to the wetland.

Stream stage was continuously recorded at two gages located at the upstream and downstream ends of the wetland. Stage-discharge relations were established at both sites based on periodic current-meter measurements of flow. The stage record at each gage and the stage-discharge relation were used to calculate streamflow. Mean daily flow data for the upstream gage were used to compute the streamflow volume entering the wetland area. The ratio of streamflow volume to drainage area at the upstream gage was used to calculate estimates of surface and ground water inputs from ungaged upland areas that drain directly into the wetland. Thus, upland inputs (ULI) are composed of both gaged and ungaged inputs; the ungaged inputs were estimated by applying the

relation between runoff and drainage area, as defined by the gaged upland area. The volume of surface runoff leaving the wetland basin (SRO) was calculated from mean daily flow gaged at the downstream limit of the wetland.

Evapotranspiration (ET) was estimated using climatological data and estimated water retention factors of the wetland soil (Thornthwaite and Mather, 1957). Because the wetland is underlain by a water retaining peat layer, and because water levels remained near the land surface during the growing season, a maximum water holding capacity value of 400 millimeters was used to calculate potential wetland evapotranspiration.

Changes in the amount of surface and groundwater stored in the wetland (ΔST) were calculated based on the net differences in surface and ground water levels observed over an annual period and the porosity of the material dewatered or recharged. For the annual budget calculations, changes in the amount of surface and ground water stored in the wetland were small relative to the flow through the wetland/stream system.

Inputs and outputs of the water balance varied seasonally. For example, upland water input represented a large part of the total input in March, 1985 (74 percent) (Fig.1), but represented only 50 percent of total input volume in July of that year (Fig.2; this occurred despite the nearly equivalent precipitation volume falling during both months (Fig.3). The higher volume of upland runoff during March occurred, in part, because precipitation stored as snow and ice in the basin during the winter was released during March when the mean monthly air temperature rose above 0 degrees centigrade. Evapotranspiration also varies seasonally. In March, 1985, evapotranspiration was estimated to be about 0 percent of the total output (Fig.1). In July, at the height of the growing season when streamflows were low, evapotranspiration was 56 percent of the total output (Fig.2). Figure 4 shows the monthly distribution of water-balance components for 1985. This type of information about the hydrologic regime of a wetland demonstrates the hydrologic response of the wetland to seasonal change, which is useful for understanding seasonal differences in streamflow quality. During periods of high flow from snowmelt, dilution of dissolved and suspended materials derived from the wetland occurs; during low-flow periods when evapotranspiration losses are significant, concentration of these materials occurs.

Comparison of monthly values for a single water balance component over the annual period also demonstrates the hydrologic response of the wetland to seasonal change. For example, a large precipitation volume occurred in August, 1985 (Fig.3), but was not reflected in a correspondingly high surface runoff output volume for the wetland (Fig.5). During that period, part of the precipitation replenished soil moisture. Figure 6 shows that evapotranspiration during August and July had been higher than other months of the year; evapotranspiration outputs partially deplete soil moisture during low flow periods. Data show that wetland ground water levels were at a minimum during July, and began to rise in August (Fig.7), which illustrates the replenishment of soil moisture in August.

Comparison of precipitation (Fig.3) and surface runoff output (Fig.5) in September and October 1985 indicates that, although the volume of water from precipitation was nearly equal in both months, surface runoff from the wetland in October was significantly greater than in September. The reduction in evapotranspiration (Fig.6) during October following senescence accounts for much of the increased volume of runoff. This information is useful for understanding seasonal differences in stream water quality related to seasonal changes in the water balance. For example, the quality of stream water during this seasonal period is not likely to be characterized by evapotranspiration-induced concentration of dissolved and suspended materials.

The hydrologic regime of a wetland can vary from year to year. Surface runoff output from the wetland in 1985 was distinct in several ways from that in 1986. During the first half of 1986, surface runoff volumes leaving the wetland were 60 percent greater than in 1985 (Figs. 5 and 8). This occurred despite the equivalent annual precipitation volume that fell on the wetland in 1985 and 1986 (1.99 and 2.01 million cubic meters, respectively). The volume and timing of individual precipitation events in late 1985 and early 1986 account for the increased runoff volume that occurred in 1986. From March to August 1985, there was a steady decrease in surface runoff out of the wetland (Fig.5). Then, in response to the large volume of precipitation in August (Fig.3), surface runoff volumes remained higher than they had been prior to the August precipitation. Ground water levels also increased during that period (Fig.7), indicating that ground water recharge had occurred. Surface runoff volumes and ground water levels remained high through April 1986 (Figs. 8 and 9). Then, during the growing season in 1986, surface runoff and groundwater storage declined to levels similar to those prior to the August 1985 precipitation. This illustrates the importance of summer rainstorms in replenishing soil moisture or recharging ground water; the large precipitation volume that fell in August 1985 effectively recharged ground water storage so that runoff volumes increased by 60 percent from August through the first half of the following year in response to monthly precipitation inputs.

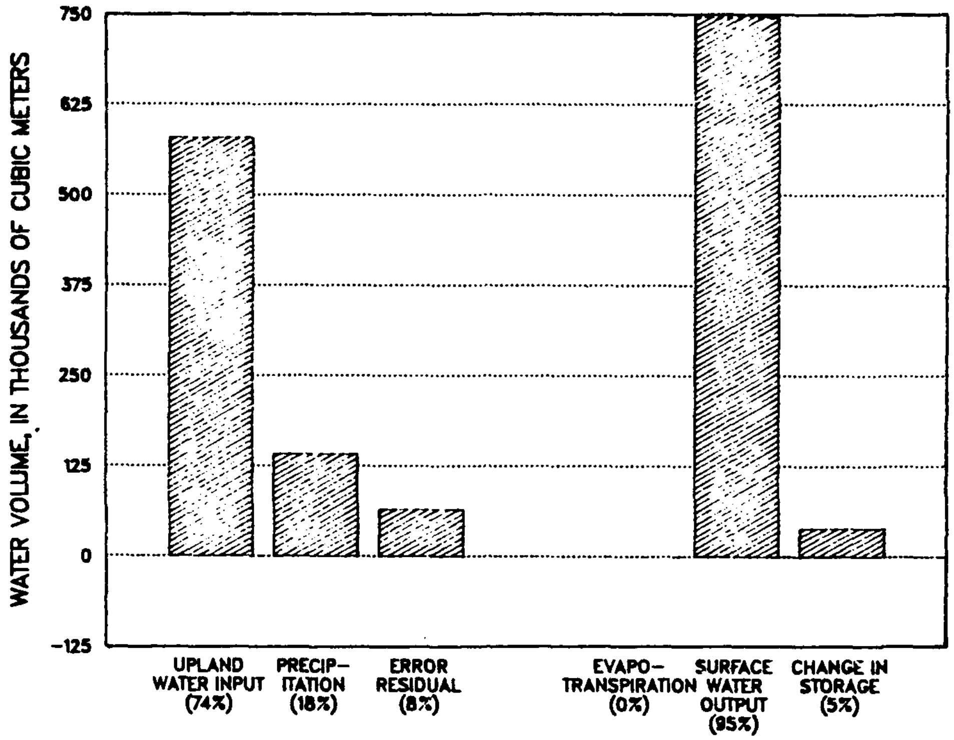

Figure 1.--Inputs and outputs of water for wetland, March 1-31, 1985.

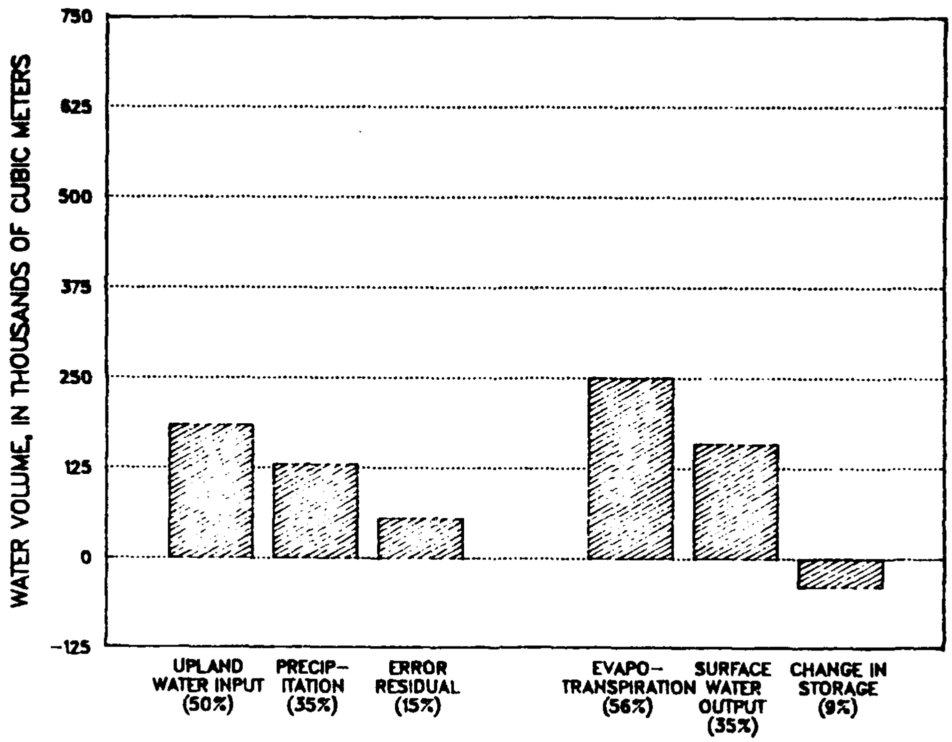

Figure 2.--Inputs and outputs of water for wetland, July 1-31, 1985.

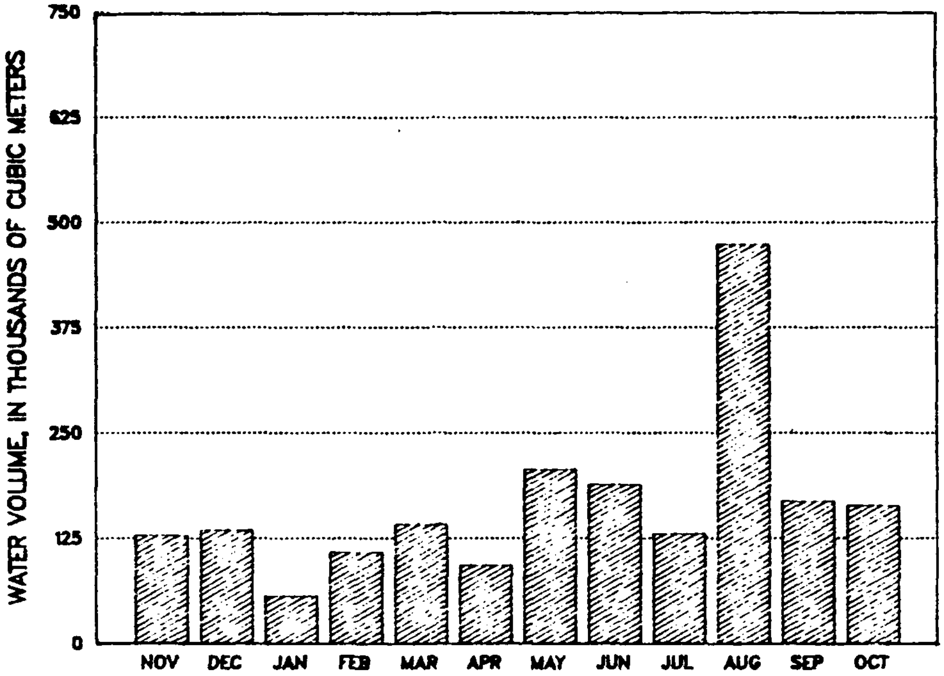

Figure 3.---Precipitation input (P) to wetland, 1985.

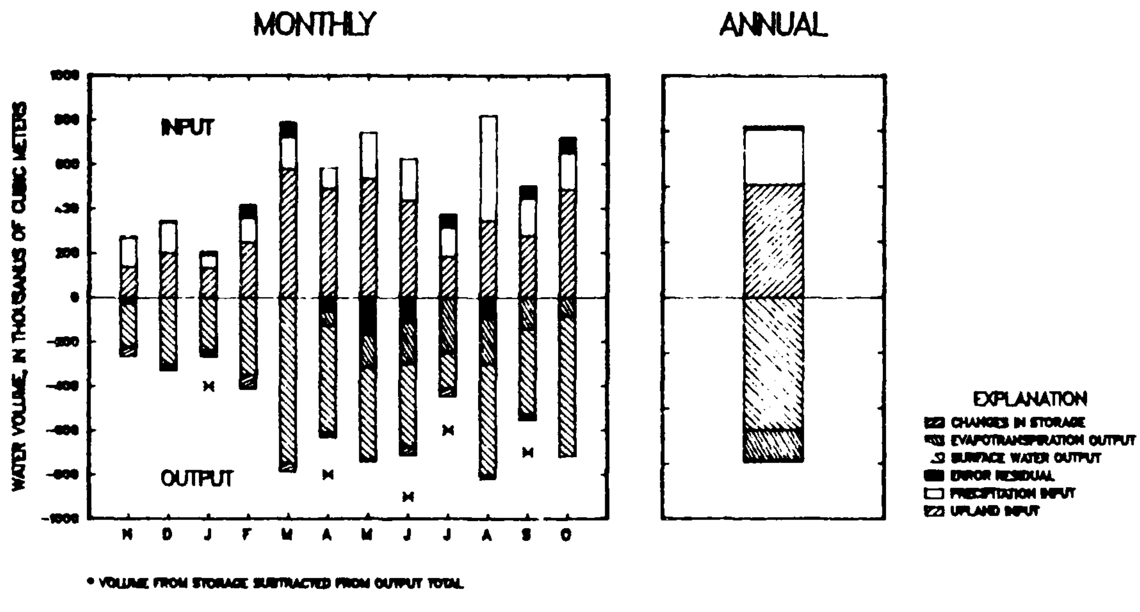

Figure 4.---Monthly distribution of water-balance components, 1985.

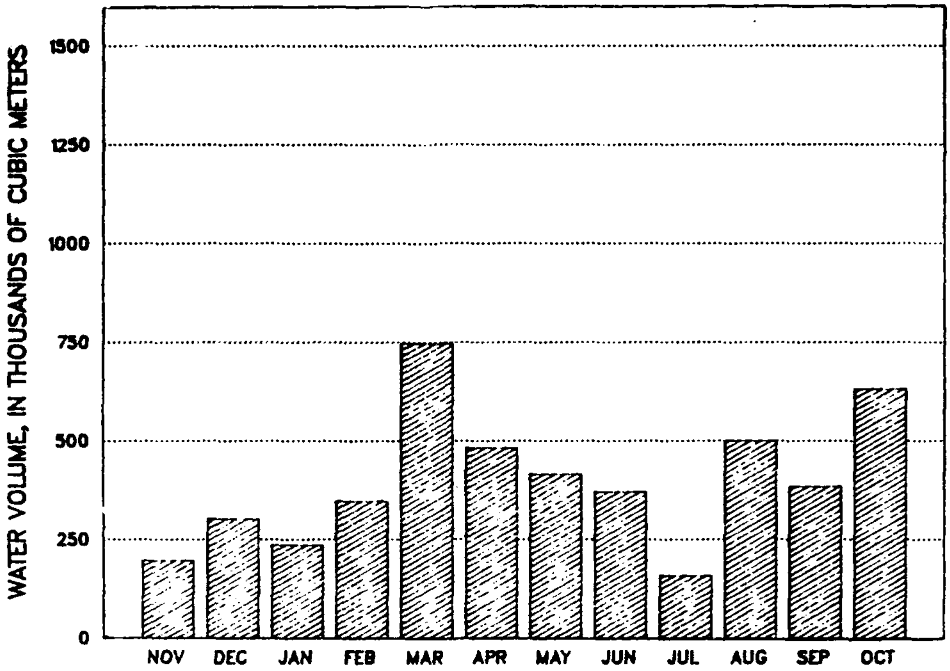

FIGURE 5.--Surface runoff output (SRO) from wetland, 1985.

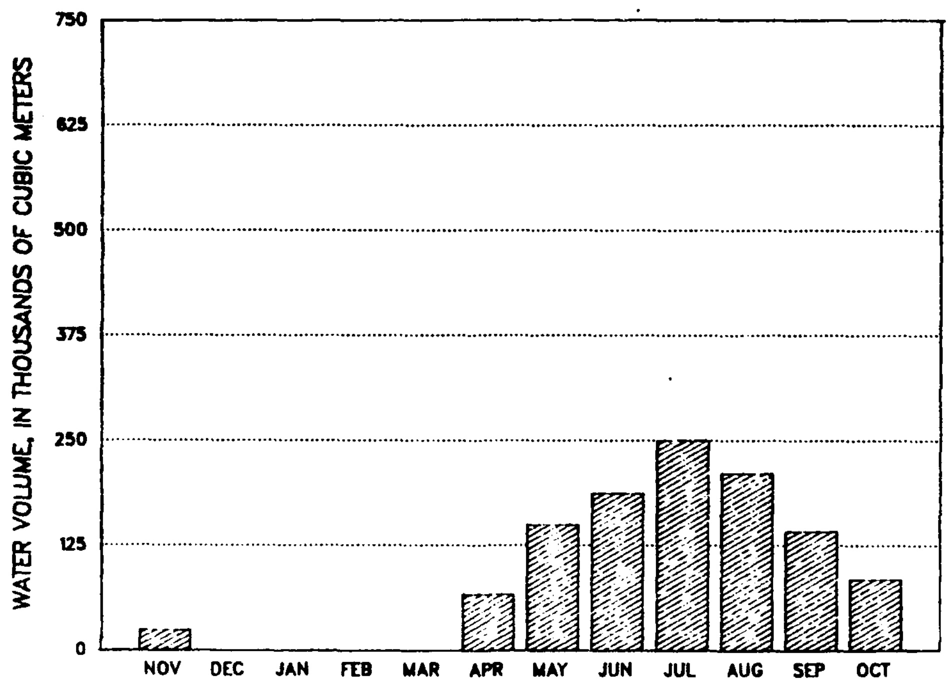

Figure 6.--Evapotranspiration output (ET) from wetland, 1985.

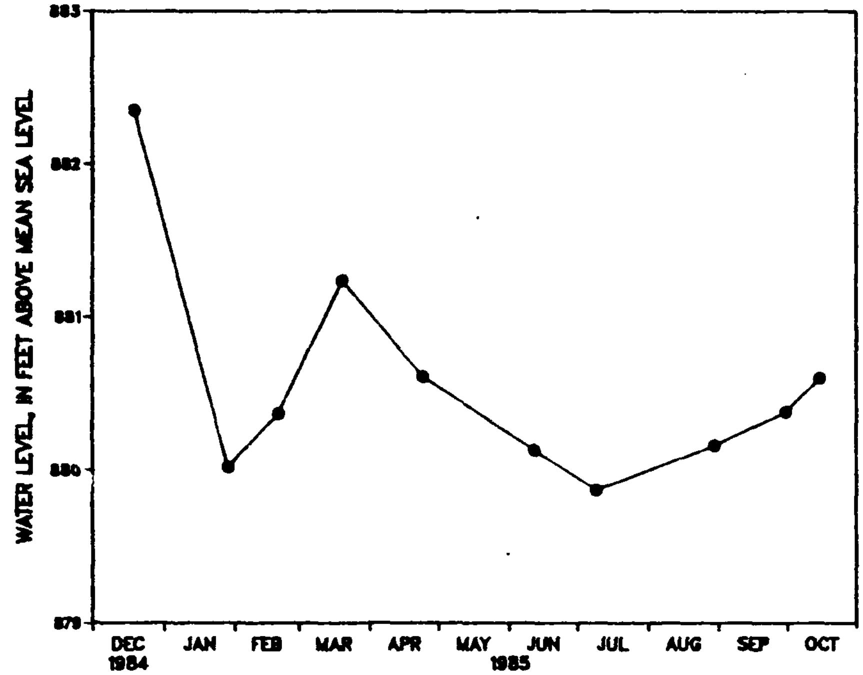

Figure 7.—–Water levels in wetland observation well, December 1984 through October 1985.

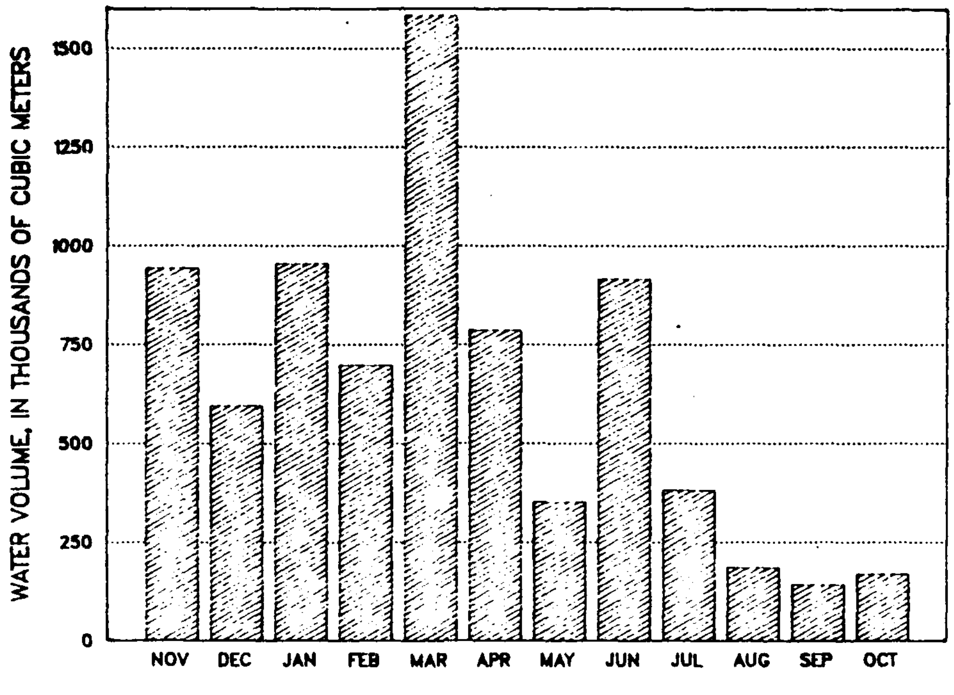

Figure 8.—–Surface runoff output (SRO) from wetland, 1986.

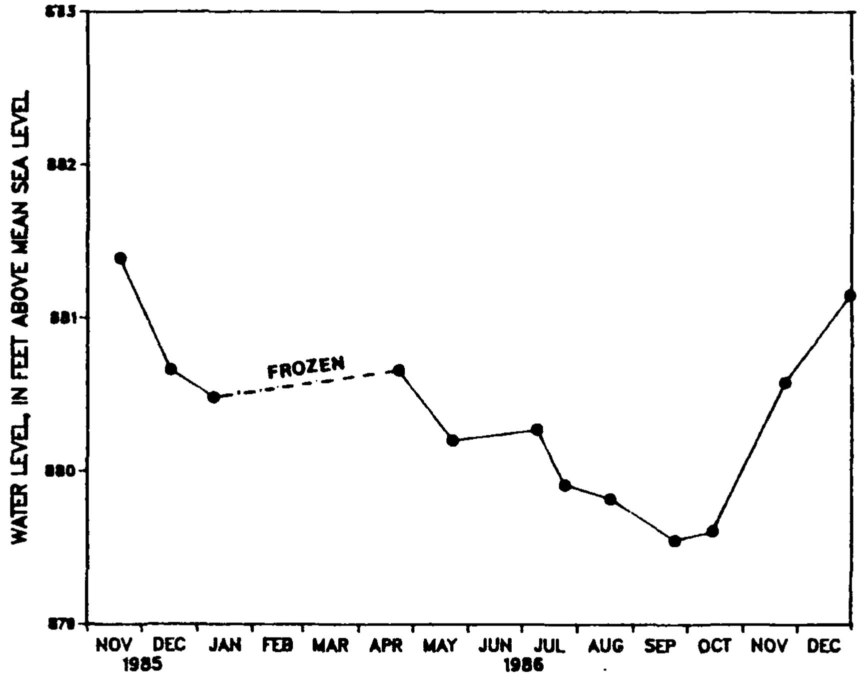

Figure 9.--Water levels in wetland observation well, November 1985 through December 1986.

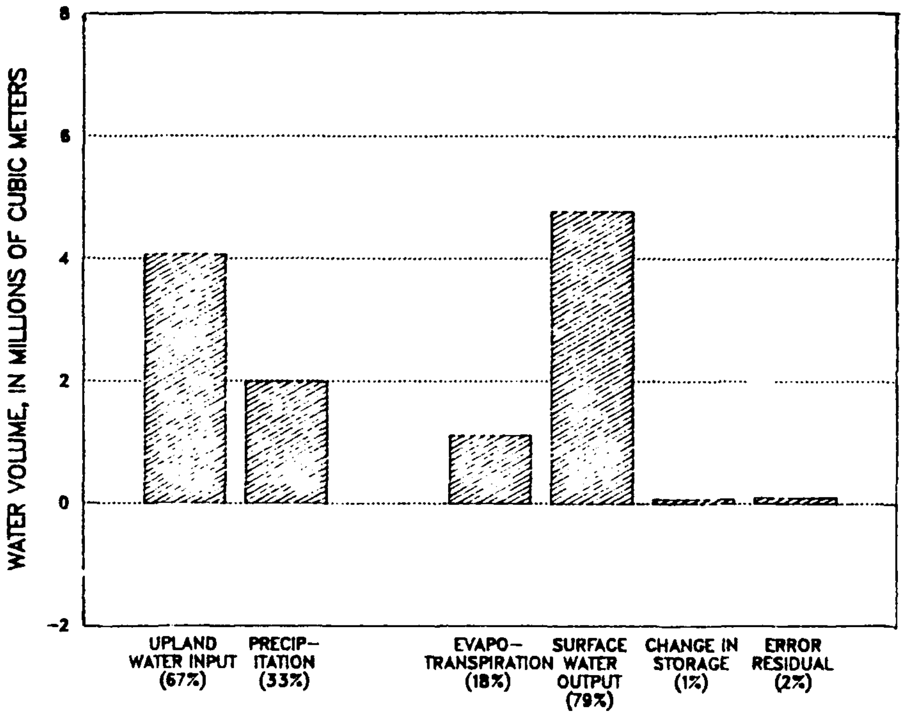

Figure 10.--Inputs and outputs of water for wetland, November 1, 1984 through October 31, 1985.

METHOD LIMITATIONS

Uncertainties are inherent in the measurements of hydrologic and physical characteristics of any environmental system. In an effort to quantify error of the water balance, estimates of measurement error were made for each method of hydrologic data collection, and a percent error was assigned to each water balance component. Measurement error was estimated to be 20 percent for upland water input, 10 percent for precipitation, 10 percent for surface runoff output, 15 percent for evapotranspiration, and 50 percent for changes in ground and surface water storage. However, total error in the water balance cannot be calculated merely by summation of all component errors, because it is never known whether each individual error is positive or negative with respect to the "true" value. In other words, accumulation of error probably is not in the same sign direction, and, therefore, errors tend to compensate. For the annual water balances (Figs. 10 and 11), the difference between inputs and outputs (error residual) is 2 percent for 1985 and 5 percent for 1986, even though the estimated error for any single water balance component is at least 10 percent. Error compensation becomes less profound as shorter time periods are used in the water balance. Most monthly water balances had residuals that were less than 10 percent of the input or output. However, several of the monthly water balances had residuals greater than 10 percent of the input or output. These larger residuals are likely the result of net hydrologic measurement error and some balance component volume that was not completely accounted for, such as ground water flow.

The residual term of the annual water balance is small relative to the potential measurement error. Therefore, it is difficult to determine how much of the residual represents water inputs or losses that were not completely accounted for and how much is an artifact of the large potential error of the balance. Error and uncertainty such as this makes subsequent interpretation more difficult. For example, interpretation of chemical budgets based on an inexact quantification of the hydrologic regime becomes even more uncertain. Error in mass balances can exceed the error of the water balance, particularly if the water balance component that accounts for most of the loading

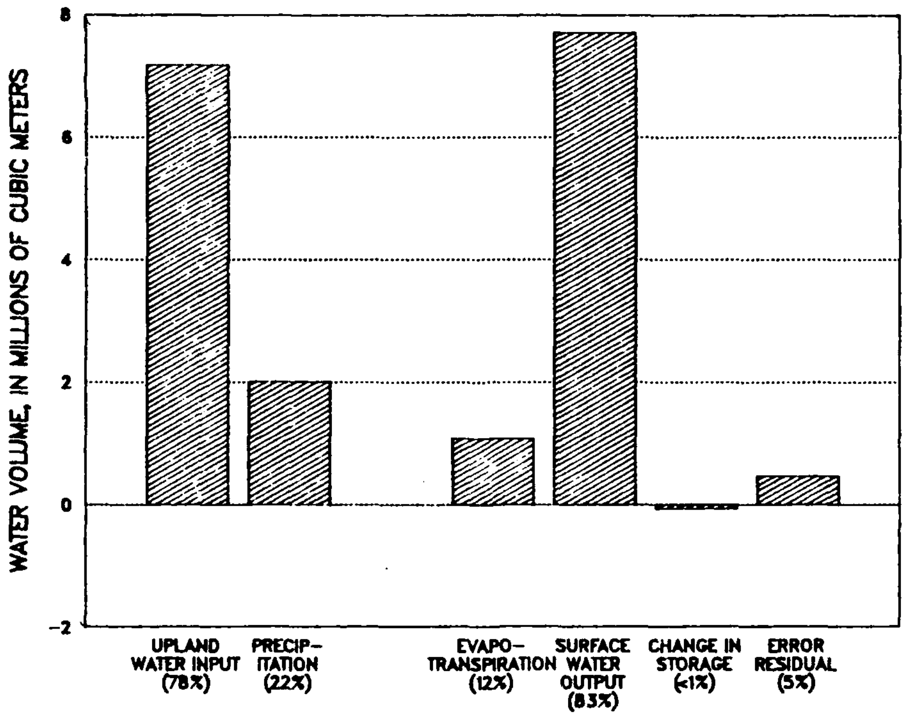

Figure 11.—Inputs and outputs of water for wetland, November 1, 1985 through October 31, 1986.

has a large error associated with it.

SUMMARY AND RESEARCH NEEDS

Despite the inherent uncertainty in hydrologic and physical measurements, this water balance method is a useful way to gain an understanding of the hydrodynamics of a system that in turn control the nutrient storage and cycling function of wetlands. The critical hydrologic elements of a wetland system can be identified, as can the relative importance of each element as a possible source or sink of water quality constituents under varying hydrologic and seasonal conditions.

Wetland research needs to be prioritized to include studies that take a holistic approach to understanding the hydrodynamics of wetlands. As new information is obtained about these complex environmental systems, improvements in the methods used to quantify hydrologic and physical characteristics will result.

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chapter four

Hydrologic Changes and Wetlands

Hydrology as an Index for Cumulative Impact Studies

Nancy C. Taylor Leibowitz, Roel Boumans, and James G. Gosselink
Center for Wetland Resources
Louisiana State University

INTRODUCTION

Hydrology is deemed to be the most important factor controlling wetland structure and function (Taylor et al, 1984). "The source, velocity, renewal rate, and timing of water in a wetland ecosystem directly controls the spatial heterogeneity of nutrients, O_2, and toxin loads in sediments" (Gosselink and Turner, 1978). Wetland primary productivity, wildlife habitat, species composition and diversity, nutrient cycling, organic deposition and flux, heritage, harvest, and aesthetics are all tied to the presence, movement, and quality and quantity of water (Carter et al, 1979; Gosselink and Turner, 1978).

Wetlands, in turn, moderate water flow. Hydrologic values of forested wetlands include low flow augmentation, flood storage and control, and deep aquifer recharge (Taylor et al, 1984). Wetlands store water during severe floods and release it during dry periods to mediate flows (Carter et al, 1979). Forested wetlands reduce stage heights and water velocities downstream by prolonging the flood period at lower levels.

Because of this close interdependence of wetlands and water flow, hydrologic cumulative impacts significantly affect the way wetlands function. (Cumulative impacts are the sum total of incremental changes occurring through time; each individual change is inconsequential, yet the total change results in significant ecological impacts to an ecosystem.)

Hydrologic changes in a watershed occur in a number of ways. Structural disturbances in a watershed often result in dramatic hydrologic responses. Channelization affects the duration and stage of downstream flooding by moving the water more rapidly through the channelized reach (Lavine et al, 1974; Kuenzler, 1976; Wharton et al, 1982). As an example, Belt (1975) illustrated that an aftermath of the construction of major flood control projects on the Mississippi River was that the average river stage for a given flood discharge had increased. Conversely, during low discharge intervals, the river stages were lower than expected from the historical record, because the leveed reaches were now more efficient at moving waters. In addition, the levees along the river banks cut off the return flow of backwater which would have reached the stream during low flows.

Grade stabilization structures and weirs also cause decreased stages and discharges below these structures (Simon, 1976). The obstructions lessen the effective hydraulic head and reduce water levels downstream. At lower stages, viscous drag significantly influences flow over board crested weirs, and a boundary layer forms in the velocity profile of the overflow. In addition, if free fall over the weir is prevented by the downstream stage, the discharge coefficient is reduced and thus the discharge is decreased.

Deforestation and agricultural leveling also result in destabilization of hydrology within a watershed. Deforestation reduces surface friction and increases rates of runoff and erosion. In addition, agricultural leveling of the deforested land reduces storage capacity. The elimination of the forested wetland's capacity to store backwater floods results in higher stages and discharges for a given flood (Wharton et al, 1981). Conversely, water stages and discharges are lower during times of low flow, because of the decreased seepage previously supplied by the forested floodplains.

OBJECTIVES

The objective of this study was to determine the cumulative impacts on the hydrology of the Tensas River bottomland watershed, including both impacts due to deforestation as well as those caused by structural modifications of basin streams. Historical data, including discharge measurements and stage heights, were related to previous flood control construction and bottomland hardwood clearing. More specifically, to accomplish this objective we:

1) Plotted stage heights and calculated discharges through time for Bayou Boeuf at Girard.

2) Compared these historical changes with the timing of deforestation and structural modifications (channelization, stabilization, and weirs).

STATION DESCRIPTIONS

Based on the availability and longevity of data within the boundaries of the Tensas River Watershed, three hydrologic stations were chosen: 1) The Boeuf River at Girard, 2) Bayou Macon at Delhi, and 3) the Tensas River at Tendal (see Figure 1). All of these stations included stage discharge records from the late 1930's to the present and were located along the east-west Illinois Central Gulf Railroad which bisects the basin. Because of time constraints, our analysis discusses only the results from the hydrologic station on Bayou Boeuf at Girard.

Boeuf River at Girard was the western station of the three, located at Latitude 32°28' 52", Longitude 91°47' 52". The gauge was situated on the downstream side of a bridge on U.S. Highway 80, 0.5 miles west of Girard. The drainage area of this station included 1,226 mi² in Morehouse, West Carroll, and Richland Parishes. The period of

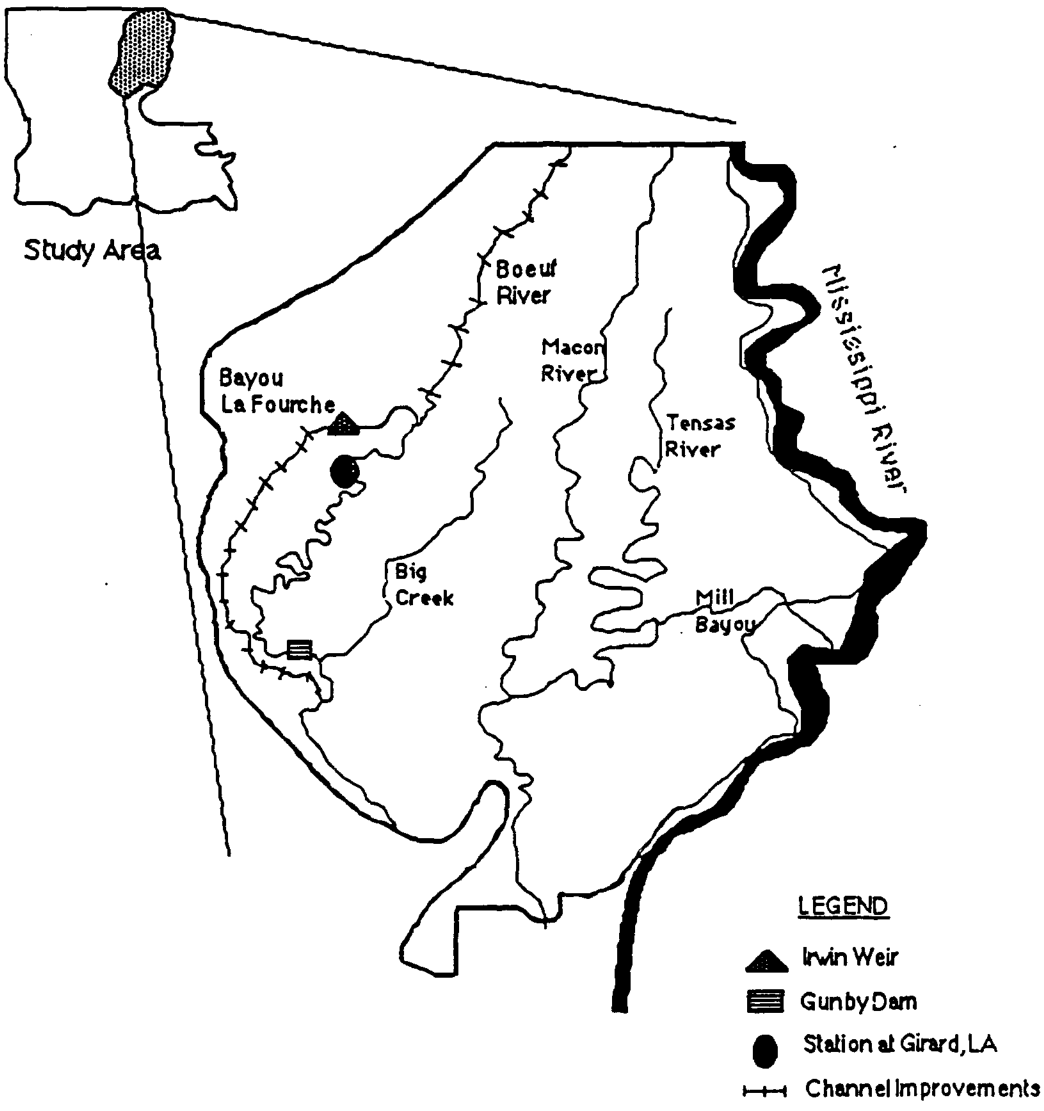

Figure 1. Major Streams and Structural Changes affecting the Boeuf River, Tensas Basin, North Louisiana.

record for this station was Oct. 1938 through the current year. Remarks in Carlson et al (1983) indicated that large diversions above the station were made for irrigation purposes. Between 1949 and 1960, major channelization and dredging activities (including placement of spoil on channel banks) took place in the upper Boeuf River. This activity affected the hydrology of 85,511 acres in Morehouse, 56,193 acres in West Carroll, and 94,934 acres in Richland Parishes. In addition, the Boeuf River diversion in Bayou Lafourche for flood control was completed in the early 1950s. Structural changes, including enlargement of the lower 39 miles of Bayou Lafourche, in combination with the Irwin Weir, insured a 90:10 percent discharge split respectfully between the Bayou Lafourche and the Boeuf River. Thereafter, sediment aggradation in the Boeuf River caused it to revegetate and further reduce its flow to 5% of the original discharge.

METHODS

Stage, calculated discharge, and stage-discharge rating curves were obtained from the United States Geological Survey (USGS) for the predesignated stations. The slope value, in the USGS-calculated stage versus discharge curve for every available year, was calculated between the 9' and 13' stages (see Figure 2). The selected stage levels were at the upper end of the stage-discharge rating curves and were indicative of discharge measurements when overbank flooding occurred. The stage/discharge ratios were plotted as suggested by Gosselink and Lee (1987).

Trends in both stage and discharge through time were scrutinized for the Boeuf River station using each of the following procedures:

1) The minimum stages and discharges of each year in the record were plotted.

2) The maximum stages and discharges of each year in the record were plotted.

3) The observed changes in the hydrographs were compared with the 1983 LANDSAT aerial imagery.

The structural effects of channelization, weirs, and excavation within each watershed were obtained using information in McDonald et al (1979), Stavins (unpub. manu.), Smith (unpub. manu.), and COE and SCS project maps. Structural effects upstream were included, but structural effects further than approximately 10 miles downstream from the station were excluded from analysis (C. Thomas, pers. comm., COE, Vicksburg, Miss. 39108-0060). The combined analysis of the trends in the two comparisons, and their relation to the historical background of the area, led to indications of cumulative impacts in the region.

RESULTS

Stage/Discharge Ratios

The stage/discharge ratios in Figures 2 (a), (b) and (c) illustrate different trends in efficiency for the three sites. The Boeuf River rises faster in 1985 than in 1935 for a given increase in discharge, that is, the stream has become less efficient. The Macon River, after 1960, shows a decrease in stage for a given discharge through time. It has become more efficient. The Tensas River shows no consistent change in slope through time, although there has been considerable fluctuation through the years.

Minimum Stage and Discharge at Bayou Boeuf

Simultaneous comparison of the minimum stages and minimum discharges on the Boeuf River illustrate opposite trends (see Figure 3). The minimum stage, low and stable during the first several decades, rose dramatically after 1965 and became slightly erratic. The minimum discharge was erratic and high during the first several decades, and decreased drastically, to less than 20 cfs after 1965, becoming relatively stable.

Maximum Stage and Discharge at Bayou Boeuf

Comparisons of the maximum stages and discharges through time reveal trends similar to minimum stages and discharges (Figure 4). The maximum stages were low and stable during the first several decades, whereas after 1965 they became elevated. The maximum discharges were erratic and elevated during the first several decades, but they decreased to about 20-25% after 1965.

DISCUSSION

Hydrograph Alterations

Major changes have occurred in the maximum stage and discharge hydrographs at the station Bayou Boeuf at Girard. Comparing the 1940 to the 1955 period to later years, the maximum stage increased from 20' to about 23', the minimum stage increased from 0' to 6', the minimum discharge decreased from 100 cfs to 10 cfs, and the maximum discharge decreased from 3000 cfs to an average of 1800 cfs. The stage/discharge ratio has increased, indicating flashier flooding events.

Both the maximum and minimum stages were relatively stable during the period from 1937 to 1960+, yet they experienced increased levels after the completion of the Bayou LaFourche Diversion. Both minimum and maximum stages, which are currently high and stable, are a result of the decreasing slope of the Boeuf River, caused

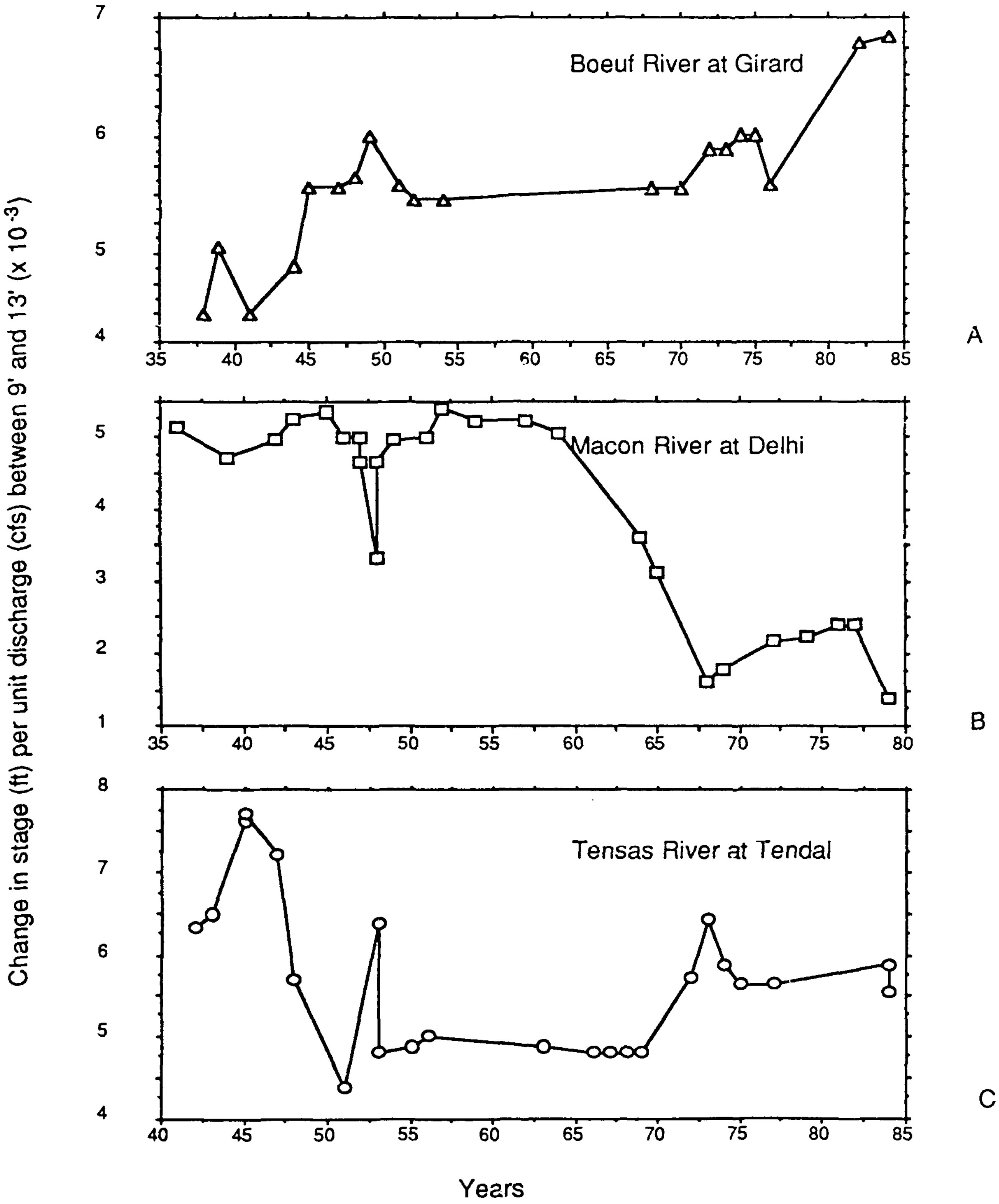

Figure 2. Slopes of discharge Rating curves between 9' and 13' at the different sites:
A) River Boeuf at Girard
B) Macon River at Delhi
C) Tensas River at Tendal

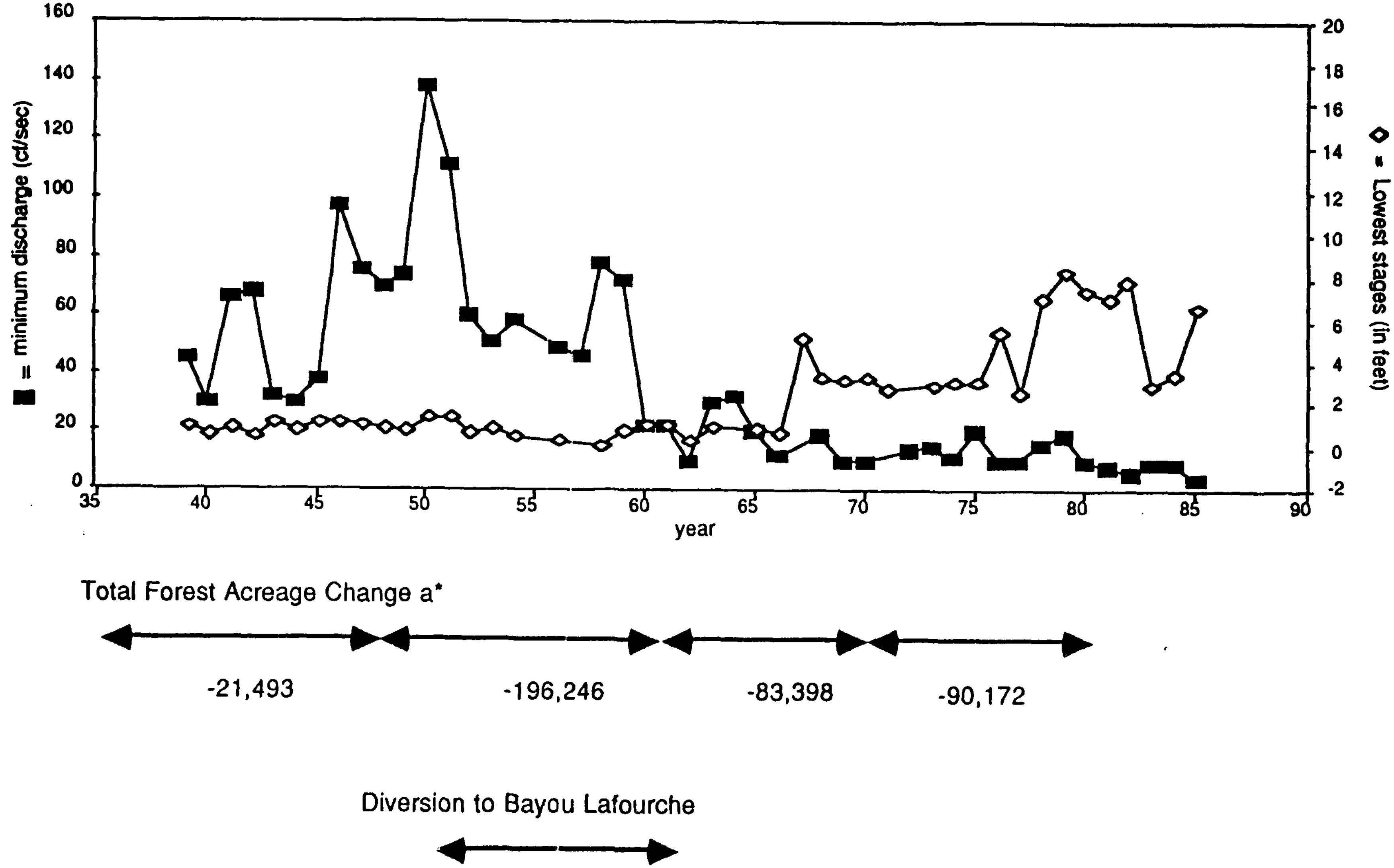

Figure 3. Minimum annual discharge and lowest annual stages versus time, structural changes to the river through time, and deforestation through time at Boeuf River at Girard.

a* =Total forested area change as reported in McDonald et al.1979 vol II, for Morehouse, West Carroll, and Richland Parishes

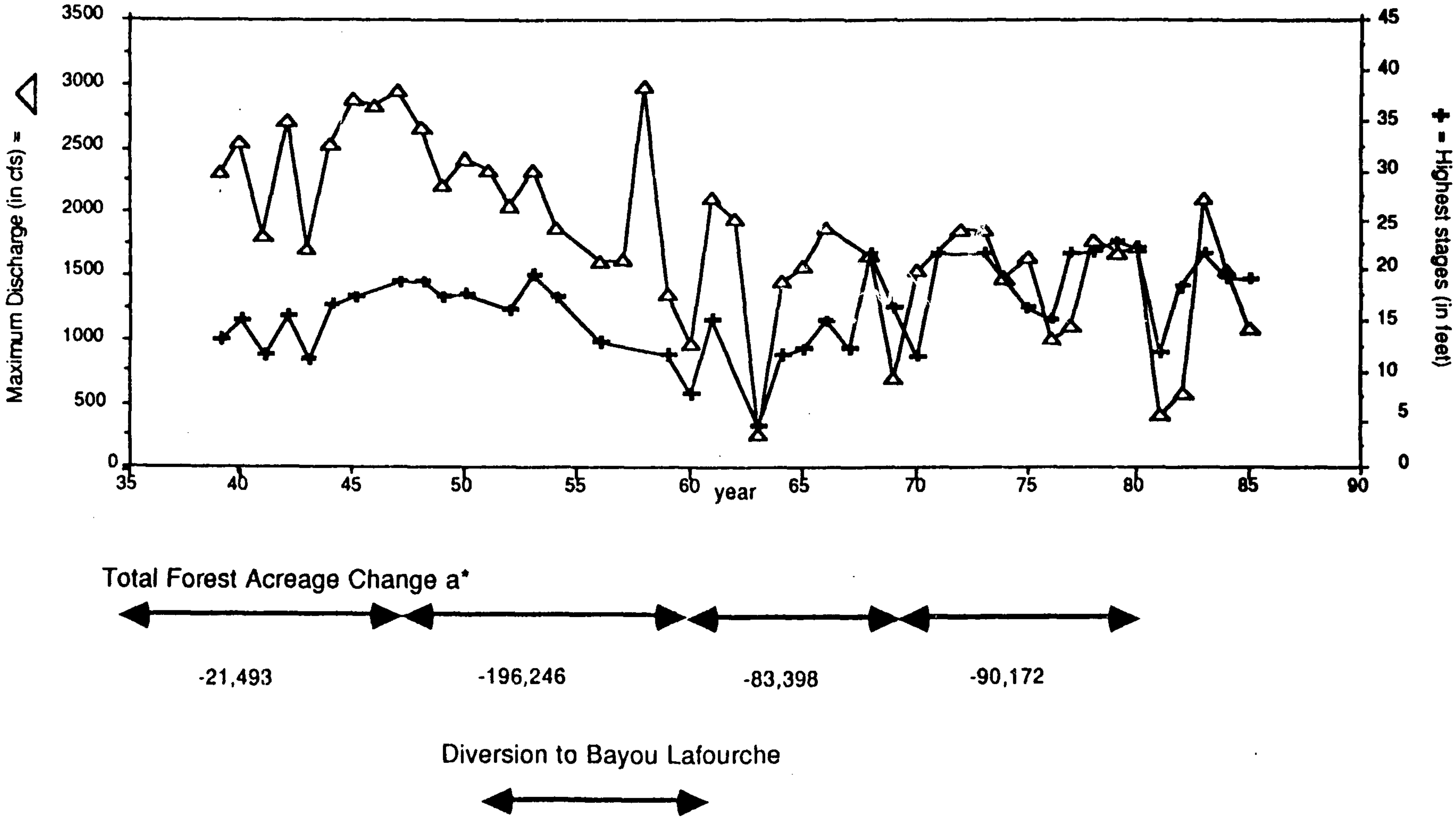

Figure 4. Plot of maximum discharge and stage heights through time, structural basin changes through time, and deforestation changes through time at Boeuf River at Girard (a* as in Figure 3).

by increased sedimentation, beaver dams, and vegetation. In addition, the Gunby dam downstream from this station may be artificially elevating the stages at this site. Some variability in the oscillating maximum stages may be due to private irrigation pumps observed in the area (T. Smith pers. comm., COE, Vicksburg District, Vicksburg, Miss. 39180-0060).

Minimum and maximum discharges were erratic and elevated between 1937 and 1960, yet they decreased somewhat after 1960. Both experienced erratic increases from 1937-1950 (1937-1947 in the case of the latter) and decreases from 1951-1960 (1947-1960 in the case of the latter). Part of this discharge fluctuation is due to climatic variability, including mean yearly rainfall differences. The erratic increase in maximum discharge from 1937-1947 may be due to deforestation in the system, since no structural changes in the basin occurred. The decreased minimum discharges in the period from 1950 to 1957 were probably due to both cumulative impacts of deforestation and the structural diversion of the Boeuf River into Bayou Lafourche. The erratic decreases in the maximum discharges from 1947-1960 may be influenced by both irrigation diversions (Carlson et al, 1983) and the Boeuf River diversion into Bayou Lafourche (Smith unpub. manu.).

From 1960 to the present, both the maximum and minimum discharge appear to be oscillating about an equilibrium point. Both remain markedly low and stable; this is probably due to severe reduction in flow caused by the Bayou Lafourche diversion (which has reduced the effective flow down Bayou Boeuf to 5% of its original flow) and private irrigation pumps observed in the area (T. Smith pers. comm., COE, Vicksburg District, Vicksburg, Miss. 39180-0060).

Ecological Consequence of Hydrograph Changes

The decreased discharge combined with higher water stages means that bottomland hardwood forests along the Boeuf River flood more deeply but less frequently than historically The Bayou Lafourche diversion, levees and channelization, and deforestational efforts have produced conditions in which the forest is flooded for shorter periods of time. The 1983 LANDSAT data similarly illustrates that little of the Tensas River Basin experienced backwater flooding during the spring flood.

Periodic flooding of bottomland hardwood forests is necessary for their maintenance as a productive, healthy ecosystem. Severe reductions of the flooding regime of the area significantly reduces the productivity of the bottomland forest (Mitch and Ewel, 1979) and any fauna which are dependent upon it for survival. Prolonged alterations of the flooding regime of a bottomland forest can cause its ultimate decline, allowing successional phases of upland vegetation to occur.

CONCLUSIONS

The combined effects of engineering structures and deforestational changes are reflected in the stage and discharge records at Bayou Boeuf at Girard. Dramatic changes in the hydrograph and flooding rates of this sub-basin have occurred: both maximum and minimum discharge of the Boeuf River have decreased and bottomland hardwood forest flooding has changed drastically. LANDSAT imagery from the peak flood period in 1983 portrayed little flooding anywhere in the basin.

The two comparisons, minimum stage-minimum discharge and maximum stage-maximum discharge, indicate that cumulative impacts occurred throughout the entire period studied (1937-1985). However, the two comparisons conflict on the possible decades of deforestational effects. The minimum stage-minimum discharge comparison indicated that deforestation altered the hydrology from 1937-1947. After 1955, the overwhelming hydrologic alterations caused by such public work projects as the Bayou Lafourche diversion make it impossible for us to separate the cumulative impacts of structures from deforestation in the same decade. Both structural effects and deforestation resulted in erratic changes to the stage and discharge.

The effect of this altered hydrology in the Boeuf-Tensas basin has been a severe decrease in the overbank flooding necessary for sustaining the health and integrity of the remaining bottomland hardwood forest. Prolonged hydrologic alteration of this ecosystem may cause it to succeed to an upland ecosystem.

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Cumulative Impacts of Historical Hydrologic Changes, Bolsa Chica Lowland, Orange County, California

Keith B. Macdonald
Michael Brandman Associates

Thomas W. Bilhorn
Earth Science Consultants

C. Robert Feldmeth
Ecological Research Services

INTRODUCTION

Located on the coast of Orange county, California, 28 miles south of downtown Los Angeles, Bolsa Chica lowland occupies the seaward end of a rivergap cut by the ancestral Santa Ana River through the uplifted coastal mesas of Bolsa Chica and Huntington Beach. The earliest detailed site survey (U.S. Coast Survey 1873-74) shows the entire lowland occupied by an estuarine salt marsh complex (approx. 800 ha or 1,980 acres) open directly to the Pacific Ocean (Figure 1). The estuary itself appears pristine, yet man-made hydrologic modifications inland from the estuary (upland irrigation and river diversions) are recorded as early as 1838. Since that time the physical and biological setting at Bolsa Chica has been extensively disturbed by man's activities (Moffatt & Nichol, 1971; Smith, 1981).

This paper describes hydrologic conditions in the lowland prior to site modifications and outlines the historical sequence of events that have, for all practical purposes, irreversibly altered lowland hydrology. Additional goals are to: identify specific, often unexpected, hydrologic changes that accompanied site modifications and demonstrate the value of historic maps, aerial photographs, and early records in establishing prior hydrological conditions. Finally, we hope to provide insights to hydrologic aspects of a 360 ha (900 acres) wetlands restoration presently being planned for Bolsa Chica.

BOLSA CHICA PRIOR TO 1899

Bolsa Chica is one of three former rivergaps that lie along the seaward edge of the 20 square-mile coastal floodplain of Orange County. Except for a single small settlement at Anaheim Landing (Seal Beach), the coastal area was virtually undeveloped prior to 1900. The Santa Ana River flowed unchecked from the Santa Ana Mountains and no jetties or breakwaters interrupted the natural coastal littoral regime. Tidal marshes occupied each rivergap and passed landward into extensive freshwater swamps. Talbert (1952) provides the following descriptions:

> "Originally, except for the Huntington Beach Mesa, the coastal area extending from the Newport Mesa to the Bolsa Chica Mesa and back into the country as far as Bolsa, a distance of about 7-1/2 miles, was considered a practically worthless swamp ("Gospel Swamp"). This area of about 30 square miles, 8,000 acres, was so full of peat springs and artesian wells which flowed the year around that it was quite inaccessible. It had a growth of willows, sycamores, tules, water modies, wild blackberry and other vines, grasses and shrubs that made an almost impenetrable thicket. Only cows made trails as they worked their way around and through the tules to graze on open patches of niggerhead and salt grass."

> "The superabundance of surface water, swamps, natural springs and artesian wells of Gospel Swamp seemed inexhaustible. Many places, among them Springdale and Fountain Valley, took their names from natural flowing springs and wells. Peat springs bubbled and boiled out of holes in the ground large enough to hold a good sized house without it touching the bottom or sides."

> "The extensive upper (Bolsa) bay was fed by a stream of fresh water called the Freeman River (Freeman Creek, formerly Bolsa Creek), a short river, but one which carried a considerable volume of water. It headed in Westminister and carried the storm drainage, peat springs, and artesian flow of water through what is known as

the old peat land section. The State, in a survey made in July 1918, found that the stream was flowing 500 inches of water (approx. 10 cfs) into the bay at the driest time of the year. Of course the volume of water was many times this amount during the rainy season."

Figure 1 illustrates this early pre-development condition. Map symbols (1873-74, Shalowitz in Moffatt & Nichol, 1971) indicate a "low water line" (dotted), the outer edge of marsh vegetation, and undefined "low marsh" (solid marsh lines) versus "high marsh" (dotted marsh lines). Brush or scrub covered sand dunes are present along the shore and willow thickets are indicated at several locations inland (Figure 1). An interior unshaded area is identified as an alkali flat; two other unshaded areas in lower Bolsa Gap are unlabelled, but correspond with dune-fields in subsequent maps (Dessery, 1907; Dillingham, 1971). Gilbert (1889) confirms that "Los Bolsas Creek" experienced year-round freshwater flows and notes a sand bottom, overlain by mud and covered with native oysters.

At least seven major drainage works were constructed inland from Bolsa Chica and Gospel Swamp between 18[illegible]8 and 1885. Bolsa Drainage Ditch (USGS, 1894) was constructed along the east side of Bolsa Chica marsh, to join a tidal channel east of Freeman Creek, during this period. Freeman Creek itself may also have been deepened and realigned at this time (Moffatt & Nichol, 1971). By 1904, more than 3,300 water wells -- more than half of them artesian, and with an average total annual withdrawal estimated at 125-200 cfs -- were reported in the Los Bolsas and adjacent Downey quadrangles (Mendenhall, 1905). Aided by a series of drier years, land clearing and drainage accelerated rapidly in the 1890s. The high watertable and fertile organic rich soils supported highly productive truck farming.

In summary, prior to 1889 the Bolsa Chica lowland:

- contained a fully tidal estuarine salt marsh complex, with a direct ocean connection
- maintained an equilibrium tidal inlet, open year-round
- received substantial freshwater input year-round
- experienced a shallow, seaward-dipping watertable and steady seaward flow of groundwater.

BOLSA CHICA GUN CLUB, 1899-1940

The year 1899 was a pivotal date for Bolsa Chica. At that time a closure was constructed across its main waterway which impounded the marsh complex behind tide gates. Again quoting Talbert (1952):

> "This section of the country along the coast between Long Beach and Newport Beach, south of Westminster was one of the greatest natural habitats for wild life and game birds in the world. Wild ducks, geese, jack-snipe, coots, plover, doves, killdeer, egrets, herons, gulls, pelicans, land birds and waterfowl of every kind and description varied their flights from ocean to swamp, from swamp to grain fields..."
>
> "The Bolsa Chica Gun Club was one of the first gun clubs to operate on the coast of Southern California. The standard value of land was at this time about $20 per acre. In 1895 the club applied to the state for permission to reclaim the salt water marshlands of the bay, for, at this time the tide extended far inland into the upper bay. The concession was granted under the State Tideland Overflow and Reclamation Act, which was the only way the club could acquire title to the tidelands."
>
> "The natural channel of Bolsa Chica Bay, as I have said, entered into the ocean at Los Patos. The reclamation of the tideland necessitated the closing of this channel and the cutting of a new one...connecting Bolsa Chica Bay and Anaheim Bay. Next, a dam with automatic tide gates had to be built, extending from a point of the mesa south of the club house to the sand dunes. This was rendered most difficult by the tremendous pressure of the tidal prism of water against the dam. Twice it had been washed out. I built the third dam, which is still standing with its tide gates operating...The tide gates held back the salt water when the tide was high and let out the freshwater when the tide was low, thereby keeping the salt water from going above the dam, a factor of much importance in the reclamation of the land."

'Sanding up' of Bolsa's original ocean entrance during dam construction was apparently unforeseen; an effort to reopen the ocean inlet failed and subsequently led to the canal connection into adjacent Anaheim Bay (Moffatt & Nichol, 1971). Stormwater runoff entering Bolsa Chica lowland thus now flowed to the ocean through Anaheim Bay (Lane and Hill, 1975). Tide records taken in 1932 indicate a difference of only 0.1 ft in tide heights measured at the Anaheim Bay ocean inlet and the Anaheim-Bolsa connection 3 miles upstream.

The tide gates caused several immediate

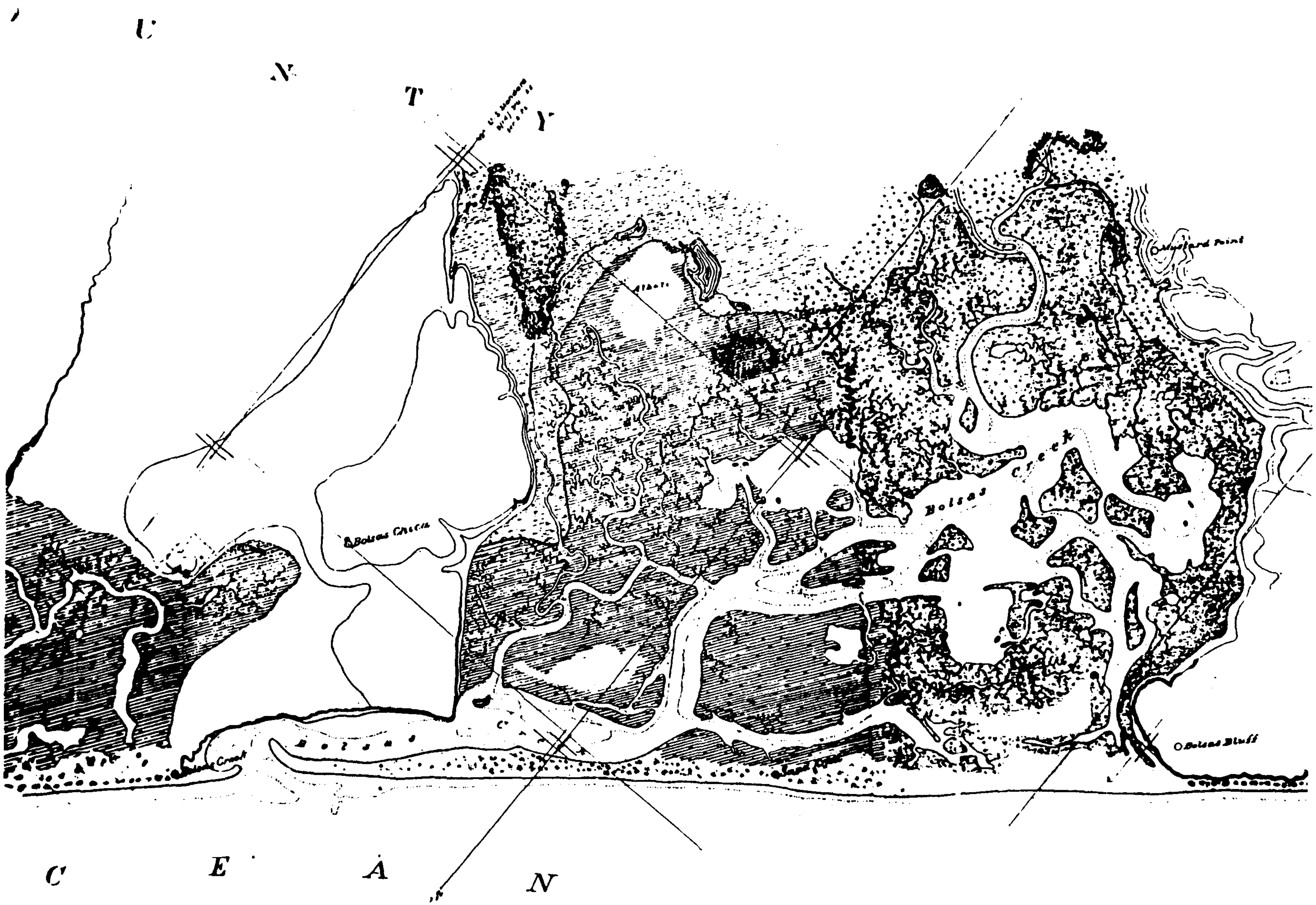

Figure 1. U.S. Coast Survey map showing Bolsa Chica estuarine wetlands complex in 1873-74. Note the extensive marsh plain, large tidal channels, and ocean inlet. The landward extension of Bolsas Creek was later renamed Freeman Creek

impacts. The prior ebb and flow of the tides across the salt marsh complex was immediately replaced by non-tidal, static water levels. Rainfall runoff and artesian flow entering Bolsa Gap no longer flowed freely with the tide, but instead ponded behind the tide gates for release at low tide. The tide gate invert at +1.5 to 2 ft MLLW was designed to maintain outflowing water levels above normal low tide. This permitted shallow-draft navigation between duck blinds, but also caused flooding upstream that resulted in two early law suits over crop damage (Moffatt & Nichol, 1971). To overcome these upland drainage problems Freeman Creek was dredged in 1908 and again in 1918-1920. New tide gates, with a lower invert elevation to allow better control over low tide outflow, were also installed by the Gun Club in 1929.

Tidal salt marsh vegetation in southern California typically extends down to MHW level (Macdonald 1977). Since the tide gates maintained water levels well below MHW:

- salt marsh submergence patterns changed from "regular-daily-tidal-marine" to "irregular-seasonal-runoff-freshwater";
- less frequent submergence promoted dewatering, compaction, and subsidence of the marsh sediments; and
- channel salinity patterns probably changed from "tidally-fluctuating/ marine-brackish" to "static/brackish-freshwater."

Many relic tidal channels and other features shown on the 1873-74 Coast Survey maps are readily identifiable in aerial photos from the 1920s and 1930s. Former channels appear dry for most of the year with freshwater flow primarily along Freeman Creek. Middle and South Bolsa Sloughs functioned primarily as overflow reservoirs during the winter rainy season (Smith, 1981).

Between 1899 and 1907, shallow duck ponds were constructed in the higher northern section of the lowland (Dessery, 1907; Dillingham, 1971); by 1919 additional ponds had been constructed in formerly tidal areas and a pumping plant supplied water for the ponds (Moffatt & Nichol, 1971). Aerial photos from 1927 show duck ponds covering most of the lowland north of Freeman Creek. To better attract migrant waterfowl, the ponds were first filled with well-water in late September or early October; winter rains then kept them supplied until early spring (Smith, 1981).

Expanding agricultural development inland from Bolsa Chica continued to deplete artesian wells and reduce Freeman Creek flow. It was during this period of decline that Talbert (1952) noted a summer creek-flow measurement of about 10 cfs (July 1918). From 1930-1940 the groundwater piezometric surface (elevation of any artesian flows) in local upper aquifers dropped below sea level during the late summer pumping cycle; a maximum lowering of 15 ft below MSL occurred in 1936.

In summary, 1899-1940 witnessed the following changes in Bolsa Chica lowland hydrology:

- Tide gates halted tidal flushing of the lowland marsh complex, but permitted discharge of inland drainage at low tide.
- Outer Bolsa Bay remained fully tidal, with access via Anaheim Bay replacing Bolsa's prior direct ocean inlet.
- Shallow, seasonally flooded, duck ponds covered up to half the former salt marsh, killing its vegetation.
- Accelerating agricultural and urban development inland lowered groundwater levels and reduced freshwater discharge into Bolsa Gap.

Littoral processes were increasingly disrupted by up-coast jetty construction (1914) and cross-gap construction of the Pacific Electric Railroad (1904) and Coast Highway, State Route 1 (1926).

OILFIELD DEVELOPMENT: POST 1940

Huntington Beach Oilfield, bordering Bolsa Chica, was discovered in 1920 and rapidly developed into the seventh largest producing field in California. A network of raised access roads and drill pads was built across the lowland (1943-1953) using fill imported from the adjacent Mesa. This network divided the lowland into a series of leveed impoundments (cells) which captured local rainfall. Culverts installed beneath selected levees maintained drainage along Freeman Creek and Middle Bolsa Slough, in the south Bolsa lowland. North of Freeman Creek, however, the individual cells were isolated from other drainages (Dillingham, 1971; Smith, 1981). Creation of the leveed cells either impeded -- as with the culvert-connected channels -- or completely halted surface flows across the lowlands. While winter runoff entering the main channels still flowed offsite via the tide-gates, temporary "seasonal ponds" were created in other cells.

By 1940, continued regional groundwater extraction caused all remaining artesian wells -- and presumably Freeman Creek -- to stop flowing. A secondary impact of prolonged oil production, along with marsh sediment dewatering and compaction, was ground subsidence. Since 1920 much of Bolsa lowland has subsided 1 to 2 ft, with estimates reaching up to 5 ft at the center of Huntington Beach Oilfield immediately east of the lowlands boundary

(Woodward-Clyde, 1984). Much of the site is presently 1 to 2.5 ft below MSL.

Expedient land disposal of oilfield brines -- by evaporation, percolation, or runoff into Bolsa Gap -- contaminated local shallow groundwater as early as 1925. Brine spread north and east from Huntington Beach Mesa out across Bolsa gap. By 1952, 95% of the brines were being pumped to the ocean. Deeper groundwater overdrafts also encouraged sea-water intrusion beneath Bolsa Gap, beginning in the late 1940's. Deliberate groundwater recharge initially halted, and then reversed, both the seawater and oilfield-brine wedges between 1962 and 1965 (Calif. Dept. Water Resources, 1968).

Planimetry of dry season aerial photos from 1928, 1947, and 1957 confirm habitat changes due to hydrologic modifications (Feldmeth, unpublished data). During the Duck Club period (1928 photos) seasonal duck pond flooding killed off former marsh vegetation. Non-vegetated flats (847 acres) exceeded vegetated marsh (338 acres) and perennial water bodies impounded behind the tide gates were minimal (47 acres). By 1947, abandonment of the duck ponds and initial creation of the elevated roadways slowed winter rainfall runoff, leading to increases in both standing water (89 acres) and vegetative cover (582 acres), at the expense of open flats (561 acres). A decade later (1957 photos) standing water had again increased (108 acres) but seasonal ponding within the bermed cells had caused die-back of vegetation (471 acres) creating additional non-vegetated flats (653 acres). Local lowland subsidence probably also contributed to the increased ponding where low-lying areas intersected the watertable.

URBANIZATION AND WETLAND PROTECTION

During the 1960s, portions of the adjacent Anaheim Bay complex were developed into Huntington Harbor, a 350 ha (860-acre) residential marina. By 1964 agricultural fields adjacent (inland) to Bolsa Chica also yielded to urban development, and before 1980 housing abutted the entire northeast boundary of the oilfield (Dillingham, 1971). Much like the earlier oilfield brines, the increased runoff and storm drainage that resulted from urban development were expediently disposed of in the adjacent lowland (Figure 2). Springdale Pump Station (emptying into Freeman Creek) and Seacliff IV Development (emptying over the edge of Huntington Mesa) each contribute to the total surface freshwater runoff entering the lowland; the remainder comes largely from in situ rainfall.

The Garden Grove-Wintersburg Flood Control Channel was constructed across the northwestern margin of Bolsa Chica lowland in 1960. It discharges into Outer Bolsa Bay through a control structure adjacent to the Gun Club tide gates. The Channel carries storm runoff from a 9,068 ha (22,400 acre) urban watershed in the cities of Garden Grove, Santa Ana, Westminster and Huntington Beach. Prior to its construction, major floods -- several of which inundated Bolsa Gap -- occurred in 1862, 1884, 1888, 1916, 1938, and 1941. Creation of the channel effectively isolated the lowlands from storm flow and sediment influx (Riznyk and Mason, 1979; Smith, 1981), however reestablishment of tidal action in the Ecological Reserve (see below) has permitted urban pollutants from the flood control channel to be diverted into Inner Bolsa Bay (Feldmeth, 1980).

Public and private interests in the Bolsa Chica lowland were the subject of a 1973 Settlement Agreement that potentially conveyed 557 acres (and certain mineral rights) to the State for wetlands restoration and a public marina, in exchange for clear title to the remaining lowland (Calif. State Resources Agency, 1972). In 1977-1978, Calif. Dept. Fish & Game diked (+4 ft MSL) off 60 ha (150 acres) of lowland obtained in the Settlement Agreement (Inner Bolsa Bay/South Bolsa Slough; Dillingham, 1971) to create a State Ecological Reserve. A muted tidal connection to the Reserve was created by opening the control gates in the old Gun Club dam (Figure 2). Groundwater levels within the Reserve prior to tidal flooding remained below sea level. A return to tidal action apparently prompted a permanent rise in underlying groundwater levels; in addition, flow restrictions resulted in tidal inundation -- both longer lasting and deeper than projected -- that killed off much of the existing salt marsh vegetation (Eilers, 1980; Feldmeth, 1980). More recently marsh vegetation within the Reserve has begun expanding in response to the new tidal regime.

Where Freeman Creek joins the Ecological Reserve, a gated control structure was constructed to allow Freeman Creek runoff to flow seaward through the Reserve levee. The structure was set too low, however, and quickly became completely blocked (Bilhorn, 1986c). Standby pumps now lift water over the levee, if needed, to protect oilfield facilities from flooding.

Construction of the Flood Control Channel and State Ecological Reserve as well as discharge of offsite stormwater into the lowland, each resulted in additional modifications to prior site hydrology (Figure 2).

Prior to completion of the Ecological Reserve levee (1978), the Gun Club tide-gates permitted low-tide outflow of winter rainfall and lowland runoff, but closed under the pressure of incoming tides. Field observations (Dillingham, 1971) revealed, however, that seawater percolated through the ocean barrier beach and bubbled up as clear "springs" along the seaward margins of Inner Bolsa Bay and South Bolsa Slough. These channels along with Middle Bolsa Slough and

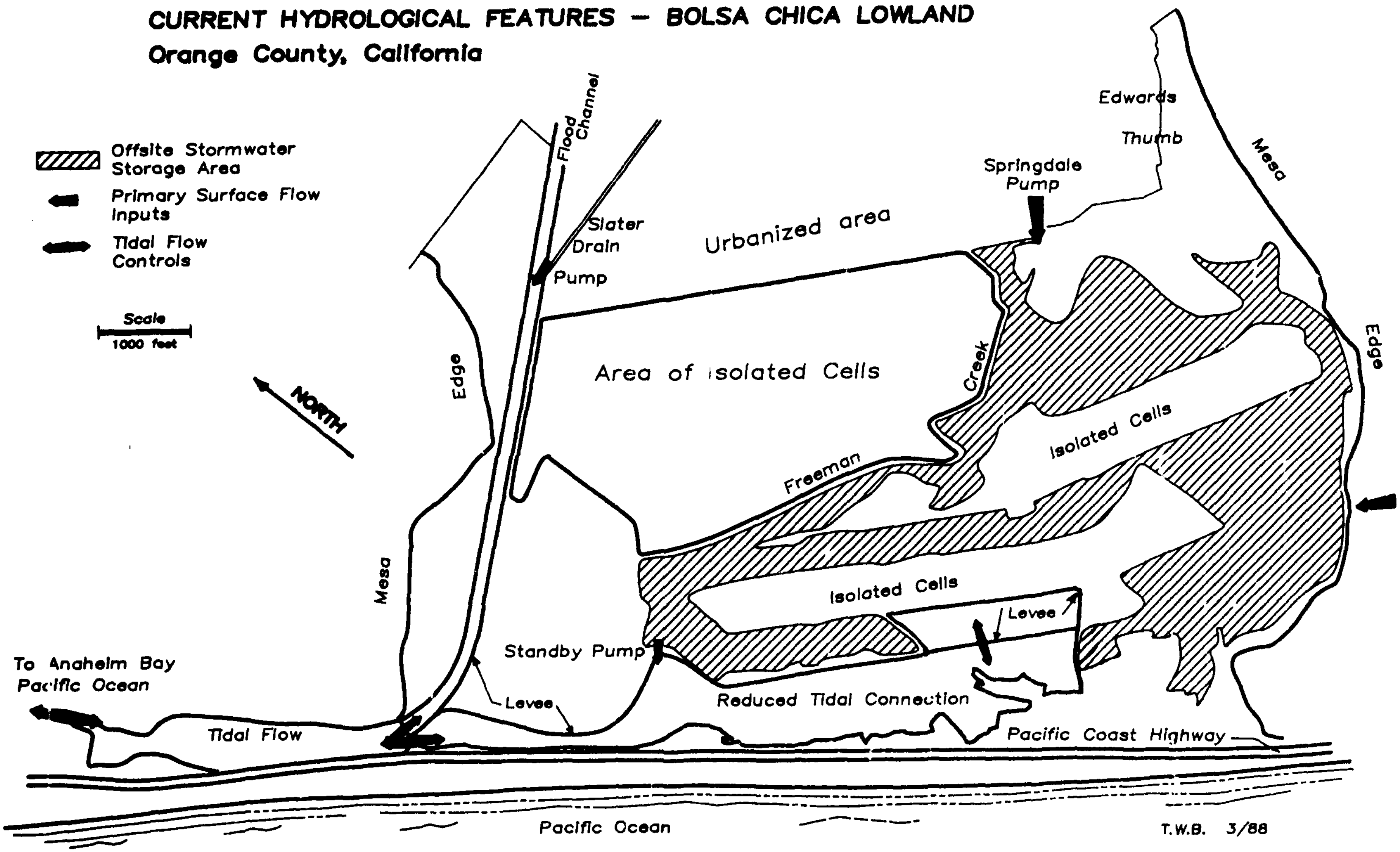

Figure 2. Present-day surface hydrological features of Bolsa Chica lowland; tidal Outer Bolsa Bay and muted-tidal Wildlife Reserve (bottom); Wintersburg Channel (left); Springdale Pump Station and Seacliff IV runoff (right) feed into interconnected cells of the stormwater storage area.

Freeman Creek always contained some seawater. A salinity gradient, increasing inland from the tidegates, was present year-round (summer 1970: 35-140 ppt; winter 1971: 36-89 ppt) and reflected slow movement of seawater inland, driven by evaporative forces and inland groundwater withdrawal (Feldmeth and Waggoner, 1972; Bilhorn, 1986b).

Following levee construction, winter runoff entering the lowland (from Springdale and Seacliff, for example) accumulates landward of the Reserve levee until it is pumped out. Winter ponding is more common than before and salinities generally lower. Since the Reserve levee now truly precludes any inflow across the lowland, inland habitats dry out more completely in the summer and fall. The net result is a more extreme physical environment than experienced in the lowland prior to levee construction.

Increasing isolation of the lowland from stormflows has disrupted natural patterns of sedimentation and nutrient influx. Zedler (1986) has also noted that prolonged stormflows ("low salinity gap") are critical to the germination and early survival of certain salt marsh plants. Indeed the loss of substantial freshwater flows and related changes in soil salinity patterns might be the single most significant environmental change experienced within Bolsa Gap.

Summarizing present day hydrological conditions at Bolsa Chica:

- A small portion of the lowland, the State Ecological Reserve, has been returned to muted tidal influence.
- Stormwater runoff from offsite urban areas enters the lowland from Springdale Pump Station and Seacliff IV Development.
- Inland from the Reserve levee, increased winter ponding and more complete drying out in the summer and fall have created a more extreme hydrological environment.
- Saline groundwater is now generally present throughout Bolsa Chica lowland as seawater moves slowly landward in response to lowered groundwater levels further inland.

FUTURE WETLAND RESTORATION

Hydrology is the controlling variable of wetland habitats; the characteristic soils, vegetation, and wildlife of wetlands develop directly in response to their unique hydrology. An appropriate hydrological regime is thus the most critical element--and in southern California probably the most difficult element--to provide for any successful wetlands restoration (Williams and Harvey, 1983; Zedler, 1986).

Returning Bolsa Chica lowland to "historical hydrological conditions" is now impractical:

- Area subsidence precludes the simple return of full tidal action--which would submerge much of the lowland and possibly adjacent housing.
- Former groundwater supplies are now overdrafted for urban uses--indeed not only has the direction of groundwater flow reversed, but groundwater salinity patterns have changed from freshwater outflow to saltwater inflow.
- The lowland is now isolated from natural stormwater flows--and surrounding urbanization precludes a reversal of this trend.

Priority should instead be given to the carefully planned reestablishment of groundwater conditions more suitable for the enhancement and creation of wetland habitats:

- With adequate water quality modifications, urban runoff from the Springdale Pump Station, Seacliff IV Development, and even Wintersburg Flood Control Channel, can partially replace the original lowland freshwater sources lost to development.
- Existing groundwater levels can be changed through sediment addition or removal to modify groundwater/root-zone relationships.
- Restoration of appropriately muted tides within diked sections of the lowland could also be designed to modify groundwater characteristics in adjacent non-tidal areas.
- Provided restoration design includes an adequate tidal prism, a direct ocean inlet to the lowland can be reestablished.
- Since both land subsidence and sea level rise will continue to influence lowland submergence, a source of sediment influx must be provided to balance these losses.

This restoration planning process for Bolsa Chica lowland is presently underway.

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Response of Mountain Stream Channels to Flow Regime Alteration

Thomas A. Wesche, Quentin D. Skinner, Steven W. Wolff, and Victor R. Hasfurther
University of Wyoming

INTRODUCTION

Over the past 20 years, the maintenance of suitable instream flows below water development projects in the western United States has been recognized as environmentally desirable and a cost developers, in many cases, must be willing to incur. Currently, one aspect of instream flows which is being actively debated by water developers and natural resource management agencies is the need for, and the determination of, flushing and channel maintenance flow requirements. Such instream flows simulate the natural spring runoff hydrograph and are felt to be necessary to maintain conveyance capacity of stream channels by reducing aggradation and encroachment of riparian vegetation, and to remove accumulated fine sediments from critical fish habitats.

Given the quantities of project water typically required for flushing/channel maintenance purposes and the associated costs of quantitative response of stream channels to flow regulation, should certain channel types respond more slowly to flow regulation, the argument can be made that the magnitude and duration of some flushing regimes can be reduced while still maintaining conveyance capacity and aquatic habitat quality.

In 1986, the Wyoming Water Research Center began a project to investigate the quantitative response of higher elevation stream channels in the central Rocky Mountains to flow depletion or augmentation resulting from water development. This project is not scheduled to be completed until late 1988, though this paper summarizes part of the project (diversions on mountain streams) and discusses some results to date.

PERTINENT LITERATURE

Although a vast amount of work has been done on changes in channel morphology due to flow regulation, the majority of that work dealt with large rivers and/or alluvial systems (Petts, 1984; Williams and Wolman, 1984). One example, Williams (1978), documented the reduction in channel size of the North Platte and Platte Rivers in Nebraska to decreases in peak discharges caused by flow regulation upstream in Wyoming and Nebraska. Very little work has been performed on higher elevation mountain streams, yet these are the systems that are currently most directly impacted by water development in the central Rocky Mountain region.

Rosgen (1985) developed a comprehensive stream classification system which categorizes various stream channels by certain morphological characteristics. Delineation criteria used in classifying channels include: 1) stream gradient, 2) sinuosity, 3) width/depth ratio, 4) channel materials, 5) entrenchment, 6) confinement and 7) soil/landform features. The primary criterion in this classification system, stream gradient, places stream channels into three classes (A, B or C). A channels are high gradient having slopes greater than four percent. B channels are moderate gradient with slopes from 1.5 to four percent. Low gradient channels, or C types, are channels with slopes less than 1.5 percent.

The two dominant forces in defining the morphological characteristics of a stream channel are flood frequencies and magnitude of the sediment load (Petts, 1984). Flow regulation of a stream system, whether by impoundment, diversion, or augmentation, will inevitably cause changes in both. Consequently, the size and shape of the channel will change. However, our ability to predict change based on the degree of flow regime alteration is very limited, as illustrated by the diversity of approaches applied and conclusions drawn by twenty professional hydrologists using three examples of reservoir and diversion projects (Simons and Milhous, 1981).

Some models that have been used to estimate morphological changes in channels under altered flow regimes include the Morphological River Model (Bettess and White, 1981), and the U.S. Army Corps of Engineer's HEC-6 (U.S. COE, 1977). Again, however, these models have primarily been applied to large, alluvial river systems. Some empirical relationships based on discharge have been derived for various stream types and systems (Leopold and Maddock, 1953; Leopold and

TABLE 1. Summary of channel response to flow depletion for mountain streams.*

SITE		WIDTH	DEPTH	AREA	W/D	C.C.
N.F. ENCAMPMENT RIVER:	Above Wolfard Canal	7.62	0.61	4.65	12.50	6.70
	Below Wolfard Canal	7.99	0.61	4.87	13.10	10.30
COW CREEK:	Above Pilson Ditches	6.04	0.76	4.60	7.92	11.92
	Below Pilson Ditches	6.49	0.46	2.97	14.20	5.11
N.F. LITTLE SNAKE:	Above Diversion	3.08	0.30	0.94	10.10	1.02
	Below Diversion (steep)	3.20	0.30	0.98	10.50	1.67
	Below Diversion (flat)	1.86	0.30	0.57	6.10	0.64
S. BRUSH CREEK:	Above Supply Canal	8.50	0.61	5.18	13.95	17.03
	Below Supply Canal	9.27	0.61	5.65	15.20	18.56
N. BRUSH CREEK:	Above Highline Ditch	9.08	0.61	5.54	14.90	8.25
	Below Highline Ditch	5.94	0.46	2.72	13.00	1.26
VASQUEZ CREEK:	Above Vasquez Diversion	8.05	0.57	4.61	14.04	9.49
	Below Vasquez Diversion	5.36	0.40	2.13	13.54	2.31
FRASER RIVER:	Above Diversion	5.36	0.46	2.45	11.73	4.74
	Below Diversion	5.52	0.39	2.17	14.03	2.89
FOOL CREEK:	Above Diversion	1.52	0.25	0.38	6.10	0.50
	Below Diversion	1.89	0.26	0.49	7.29	1.01
EAST ST. LOUIS CREEK:	Above Diversion	2.32	0.57	1.32	4.06	5.03
	Below Diversion	2.50	0.35	0.88	7.13	2.21
ST. LOUIS CREEK:	Above Diversion	5.85	0.41	2.39	14.33	4.36
	Below Diversion	6.58	0.44	2.87	15.10	5.49
WEST ST. LOUIS CREEK:	Above Diversion	2.23	0.26	0.58	8.49	1.04
	Below Diversion	1.77	0.26	0.46	6.82	0.60
LITTLE CABIN CREEK:	Above Diversion	0.67	0.25	0.17	2.65	0.28
	Below Diversion	0.61	0.22	0.13	2.82	0.17
CABIN CREEK:	Above Diversion	4.91	0.34	1.69	14.25	2.48
	Below Diversion	3.63	0.38	1.39	9.44	2.34
N.F. RANCH CREEK:	Above Diversion	3.08	0.28	0.87	10.86	0.80
	Below Diversion	2.74	0.26	0.70	10.71	1.53
M.F. RANCH CREEK:	Above Diversion	4.79	0.37	1.76	12.98	5.53
	Below Diversion	4.21	0.[illegible]1	2.55	6.93	8.83
S.F. RANCH CREEK:	Above Diversion	2.96	0.41	1.22	7.19	3.14
	Below Diversion	2.87	0.47	1.34	6.14	4.04
RANCH CREEK:	Above Diversion	3.35	0.48	1.60	7.01	7.43
	Below Diversion	3.05	0.48	1.46	6.37	5.66
LAKE FORK:	Above Homestake Tunnel	6.40	0.46	2.93	14.00	3.80
	Below Homestake Tunnel	6.95	0.55	3.81	12.67	5.54
CHAPMAN GULCH:	Above Diversion	4.30	0.37	1.58	11.65	4.03
	Below Diversion	4.11	0.39	1.59	10.63	4.72

*Width = Mean channel width (meters).
Depth = Mean channel depth (meters).
Area = Cross-sectional area of channel (square meters).
W/D = Width-Depth ratio.
C.C. = Conveyance capacity (cubic meters per second).

Miller, 1956; Simons and Milhous, 1981).

Converse to the idea of attempting to predict channel changes under altered flow regimes is the determination of flushing flow requirements (Reiser et al, 1987). Reiser et al (1987) reviewed and summarized information on existing methodologies used for recommending flushing flows and set guidelines that were determined to be necessary in the development of any formal methodology, with an emphasis on the maintenance of aquatic habitat quality in regulated systems. An example of the application of these guidelines is presented in Wesche et al, 1987.

METHODS

Work began in July of 1986 with the determination of potential sites. Selection of a particular stream for actual sampling was done onsite. Field sampling of sites was done in the summer and fall of 1986 and 1987, and consisted of sampling stream reaches immediately above and below a diversion structure. Data collected at each reach included mean channel width and depth, stream gradient, composition of the riparian zone, and composition of the streambed and banks. Several photographs (black and white prints and color slides) were taken at each site as well. All study reaches were located in the first stable, straight reach above/below the flow regulation structure which occurred out of the area of construction impact.

Based on the field data, conveyance capacity using mean channel width and depth, and channel slope was calculated for each site. Hydrologic and drainage basin data is currently being gathered and analyzed for all study reaches. Channel stability of study reaches is also being assessed using the Stream Reach Inventory/ Channel Stability Evaluation (Pfankuch, 1975).

RESULTS TO DATE

As mentioned earlier in the paper, analysis of the channel response data collected on mountain streams is not yet completed. We anticipate a project completion report will be available late in 1988. Therefore, the results presented here should be considered as preliminary and as such, will be restricted to general data trends.

Field measurements of channel width and depth were made at 39 study sites on 19 streams in northern Colorado and southern Wyoming. Site elevations ranged from approximately 2250 to 3000 m above sea level, while surveyed water surface slopes varied from less than 1.0 up to 9.8 percent. The diversion structures on the study streams ranged in age from over 100 years to less than 25 years and depleted streamflow by 5 to almost 100 percent of average annual water yield. As many of the study streams are ungaged, synthesis of discharge records is now underway. Applying Rosgen's (1985) channel typing system, 11 of the 39 sites were classified as A channels, 14 as B channels, and 14 as C.

A comparison of channel characteristics above and below the diversion structure on each stream is presented in Table 1. Response variables considered to date in our analysis include channel width, channel depth, the ratio of width to depth, cross-sectional area and channel conveyance capacity. The response of these parameters to flow depletion has been highly variable. Conveyance capacity has shown the greatest variability, ranging from a reduction of 85 percent below the diversion on North Brush Creek to an increase of 101 percent below the Fool Creek structure. Channel width was the most constant of the variables, showing a 40 percent reduction at a low gradient site below the North Fork of the Little Snake River diversion and a 24 percent widening on Fool Creek. Cross-sectional area, depth and the ratio of width to depth were intermediate in response.

The general trends of the data from Table 1 are presented in Table 2. As shown, channel shrinkage was found to occur below approximately 50 percent of the diversion structures. This phenomena was not observed at the remaining half of the study streams. It is apparent that additional analysis, taking into consideration such factors as channel slope, sediment yield, elevation, vegetation, and magnitude and duration of streamflow depletion, is needed to begin to explain the observed responses. This effort is now well underway.

CONCLUSIONS

While data analysis is not yet complete and any conclusions drawn at this time must be considered preliminary, it is quite apparent that the physical response of mountain stream channels to flow depletion is highly variable. Certain of our study streams were reduced in size due to the processes of vegetative encroachment and channel aggradation, while others exhibited no such loss of conveyance capacity. Further analysis is needed to explain this variation.

The channel maintenance issue is a complex one. Before instream flow regimes are prescribed below water development projects to preserve channel capacity and competence, it would appear that consideration should be given to the type of stream channel involved, the sediment loadings to the systems, the transporting capability of the flow regime in relation to these loadings, and the factors which govern the establishment and growth of streamside vegetation. We hope that when completed, the results of this study will help to provide some of the insight needed.

TABLE 2. Trends in channel response of twenty mountain streams in Wyoming and Colorado to flow depletion.*

	RESPONSE VARIABLE (Number of Streams)				
CHANNEL RESPONSE**	WIDTH	DEPTH	W/D	Area	C.C.
+	9	7	9	9	10
-	11	8	11	11	10
0	0	5	0	0	0
TOTAL	20	20	20	20	20

* Width = Mean channel width.
Depth = Mean channel depth.
W/D = Width-Depth ratio.
Area = Cross-sectional area of channel.
C.C. = Conveyance capacity.

** + indicates variable increased below diversion.
\- indicates variable decreased below diversion.
0 indicates no difference in variable above and below diversion.

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chapter five

Wetlands and Flooding

Evaluating Wetlands for Flood Storage

Byron Dale Simon, Lois J. Stoerzer, and Robert W. Watson
Wisconsin Department of Natural Resources

INTRODUCTION

A rapid assessment methodology was developed to assist regulators, planners, engineers and biologists to readily determine the functional significance of any given wetland to store or have the potential to store diffused surface water in a watershed. The results from this methodology provide the user with one of the following answers:

1. The wetland does not significantly contribute to flood storage within the watershed;
2. The wetland does provide significant storage for floods;
3. Additional information is needed to adequately evaluate the significance of a wetland's potential to store floodwaters.

This rapid assessment approach to analyzing the functional significance of a wetland's flood storage capacity considers the area of the wetland boundary and the average flooded depth of storage within the wetland. The average depth determination is controlled by the boundary limits of the aquatic-terrestrial vegetation intercept and its associated elevation. This will generally limit the wetland's functional significance for storing floodwaters to frequent rainfall events. As a result, the more frequent the flood event the greater the wetland's opportunity to perform flood storage and other functions such as nutrient and sediment retention.

Inversely, as the frequency of the analyzed rainfall event decreases (and flood magnitude increases), the frequency of wetland storage of floodwater and performance of other functions generally decrease. An exception to this would be when the wetland has the ability to store a large quantity of water from a major flood event. The elevation of storage for a major rainfall event is determined on the basis of the wetland boundary and not the landscape. This situation would significantly reduce downstream flood damage potential for a major flood event but is considered the exception rather than the rule in Wisconsin.

The wetland storage computation is a volumetric comparison of watershed run-off (expressed in acre-feet for a given rainfall event) to optimum or sub-optimum storage available within the wetland (expressed in acre-feet). A comparison of these two values establishes a ratio that can be used to evaluate the flood storage functional significance of a wetland. It is important, however, for the user of this rapid assessment to recognize and understand the assumptions associated with this method and the discretion imposed upon the evaluator to determine what is or is not significant and whether additional analysis is warranted.

ASSUMPTIONS WITH USE OF THIS METHOD

The assumptions for this rapid assessment include:

1. Maximum storage area of the wetland extends only to the boundary of the aquatic-terrestrial vegetation intercept.

Reason: Easiest manner to quickly define available storage. Also, limits analysis to the wetland and not to the landscape in general.

2. Flood storage analysis should attempt to match the optimum rainfall (storm) event with wetland storage to establish the highest wetland storage to runoff ratio (e.g.) 100%.

Reason: Conservative estimation of wetland significance.

3. Wetlands with a maximum wetland storage to runoff ratio of less than 25% do <u>not</u> perform a significant flood storage function.

Reason: Establishes a reasonable administrative limit for decisions.

4. Wetlands with a maximum wetland storage to runoff ratio of more than 25% perform a significant flood storage function.

Reason: Establishes decision criteria for future study.

5. Wetlands can lose up to 25% of their maximum storage capacity without significantly affecting flood storage capacity provided loss will not lower total flood storage capacity below the 25%

wetland storage to runoff ratio (see assumption 3).

Reason: Research indicates wetland encroachment of less than 25% would generally have a minimum influence on peak flows (Ogawa and Male, 1986).

6. Wetland storage and runoff are volumetric calculations only. That is to say this evaluation does not consider channel characteristics, slope, velocity, conveyance or other similar parameters evaluated in a hydraulic simulation model.

Reason: Simplification of evaluation for quick and easy decisions, while maintaining credibility by using best available techniques to estimate runoff.

MODEL

This model provides an approximate measure of the proportion of a 2 year, 24 hour rainfall event which is stored in a wetland. The model can be used for any given storm event, but for simplification purposes the 2 year, 24 hour rainfall event is incorporated into this analysis.

Information Required	*Source*
1. Watershed size at wetland outlet (acres)	USGS topo and dot grid, planimeter, or estimate by section
2. Wetland area (acres)	" "
3. Wetland flood storage depth (ft.)	Average depth of water when full to wetland boundary
4. Average runoff in 2 yr, 24 hr rainfall event (inches/acre)	Table 4 - Hydrologic soil type of watershed from Table 1. - Composite runoff curve from Table 2. - Rainfall in 2 yr, 24hr storm from Table 3.

Storage Computation

1. To obtain total runoff from the watershed in a 2 year, 24 hour storm event, multiply the average runoff in a 2 year, 24 hour storm event (inches) by the size of the watershed (acres). To convert acre-inches to acre-feet, divide by 12 inches.

2. Multiply the area of the wetland (acres) by the wetland flood storage depth (feet) to obtain total wetland storage in acre-feet.

3. Divide total wetland storage (acre-feet) by the total runoff from the watershed (acre-feet) to obtain that percent of a 2 year, 24 hour storm event stored in the wetland.

Once the wetland storage (volume) and total run-off (volume) calculations are completed the user can determine the relevance of a wetland's flood storage capacity for a given rainfall (storm) event. This assessment will also provide the user with the ability to determine the significance of flood storage losses attributed to the wetland if a portion of the wetland were filled or developed. (See sample problem)

In conclusion, this rapid assessment is designed to provide the user with a fairly rapid analytical tool to determine the flood storage significance of a wetland within a given watershed or with information as to whether or not additional analysis is warranted.

SAMPLE PROBLEM: STORM AND FLOOD WATER STORAGE CAPACITY

Project Proposal: As part of a planned airport expansion project, drainage and development of 150 acres of wetlands are proposed.

Total Drainage Area of Watershed: 3,200 acres

Total Wetland Area to be Evaluated: 240 acres

Total Wetland Acres Proposed for Drainage and Development: 150 acres

Maximum Flood Storage Capacity in Wetland: 240 acre-feet*

*Based on average depth of 1 foot.

Land Use of Watershed (use topographic maps, aerial photos, etc.)

10% Wetland
10% Residential (with average lot size being 1/4 acre)
80% Agricultural

Hydrologic Soil group D (Table 1) (based on permeability of soils and minimum rate in in/hr.)

Runoff Curve Numbers (need a composite of the 3 (Table 2)

(80%) Cultivated land with conservation treatment for group D = 81
(10%) Residential (1/4 acre lot size) = 87
(10%) Wetland = 89 (use pasture or range land - poor condition)

(.80 x 81) + (.1 x 87) + (.1 x 89) = 82 for composite curve number.

Rainfall (2 year, 24 hour storm) = 2.9 inches (Table 3)

Runoff Volume in Inches (Rainfall curve number) is between 80 and 85 so you have to average the

TABLE 1: SOIL NAMES AND HYDROLOGIC CLASSIFICATIONS IN WISCONSIN

Abscota	A
Ackmore	B
Adolph	B/D
Adrian	A/D
Ahmeek	C
Akan	B/D
Alban	B
Alcona	B
Algansee	B
Allendale	B
Allouez	B
Almena	C
Alpena	A
Alstad	C
Altdorf	D
Altoona	C
Amery	B
Angelica	B/D
Ankeny	B
Antigo	B
Arcola	C
Arenzville	B
Arland	B
Ashdale	B
Ashippun	C
Ashkum	B/D
Atterberry	B
Auburndale	B/D
Au Gres	C
Aztalan	C
Bach	B/D
Baraboo	B
Barrington	B
Barronett	B/D
Berry	B/D
Basco	C
Batavia	B
Beecher	C
Bellevue	B
Bergland	D
Bertrand	B
Beseman	A/D
Bevent	A
*Bibon	A
Billett	B
Blomford	B/D
Blount	C
Bluffton	C/D
Boaz	C
Bohemian	B
Bonduel	C
Boone	A
Boots	A/D
Borth	C
Boyer	B
Braham	B
Brems	A
Brevort	B/D
Brickton	C
Briggsville	C
Brill	B
Brimley	B
Brodale	C
*Brokaw	
Brookston	B/D
Brophy	A/D
Bruce	B/D
Burkhardt	B
Cable	B/D
Cadiz	B
Calamine	D
Campia	B
Carbondale	A/D
Carlisle	A/D
Carlos	A/D
Caron	A/D
Caryville	B
Casco	B
Cathro	A/D
Ceresco	B
Channahon	D
Charlevoix	B
Chaseburg	B
Chelsea	A
Chetek	B
Chippeny	D
Clifford	C
Cloquet	B
Clyde	B/D
Coffeen	B
Coloma	A
Colwood	B/D
Comstock	C
Conover	C
Cormant	A/D
Cosad	C
Council	B
Cromwell	A
Croswell	A
Crystal Lake	B
Curran	C
Cushing	B
Dakota	B
Dalbo	B
Dancy	B/D
Darroch	C
Dawson	A/D
Deford	A/D
Dells	C
Del Ray	B/D
Delton	B
Denrock	D
*DePere	
Derinda	C
Detour	B
Dickinson	B
Dickman	A
*Dillon	D
Dodge	B
Dodgeville	B
Dolph	C
Dorchester	B
Dorerton	B
Downs	B
Dresden	B
Drummer	B/D
Dryburg	B
Dubuque	B
Duel	A
Duelm	A
Duluth	B
Dunbarton	B
Dunnville	B
Durand	B
Dusler	C
East Lake	A
Eau Pleine	B
*Edith	A
Edmund	D
Edwards	B/D
Elburn	B
Elderon	B
Eleroy	B
Eleva	B
Elkmound	D
Elliott	C
Elm Lake	A/D
Elvers	B/D
Emmert	A
Emmet	B
Ensley	B/D
Ensign	D
Etter	B
Ettrick	B/D
Fabius	B
Fairchild	C
Fairport	C
Fall Creek	C
Fayette	B
Fence	B
Fenwood	B
*Fifield	
Fisk	B
Flagg	B
Flambeau	
Floyd	B
Forada	B/D
Fordum	D
Fox	B
Freeon	B
Freer	C
Friendship	A
Friesland	B
Gaastra	C
Gale	B
Gerwin	B/D
*Gaslyn	
Gay	B/D
Gilford	B/D
Glendora	A/D
Gogebic	B
Goodman	B
Gotham	A
Granby	A/D
Graycalm	A
Grayling	A
Grays	B
Greenwood	A/D
Grellton	B
Griswold	B
Guenther	B
Haider	C
Hatley	C
Hayfield	B
Hebron	B
Hennepin	B
*Hertel	
Hesch	B
Hessel	B/D
Hibbing	C
Hiles	B
*Hillcrest	
Hitt	B
Hixton	B
Hochheim	B
Hoopeston	B
Hortonville	B
Houghton	A/D
Hubbard	A
Humbird	B
Huntsville	B
Impact	A
Ingalls	B
Ionia	B
Iosco	B
Iron River	B
Isan	A/D
Isanti	A/D
Ishpeming	A
Jackson	B
Jasper	B
Jewett	B
Joy	B
Juda	B
Judson	B
Juneau	B
Kalkaska	A
Kane	B
Karlin	A
Karlsburg	B
Kato	B/D
Kaukauna	C
Kegonsa	B
Keltner	B
Kendall	B
Kennan	B
Keowns	B/D
Kert	C
Kewaunee	C
Keweenaw	A
Kibbie	B
Kickapoo	B
Kidder	B
Kingsville	A/D
Kinross	A/D
Kiva	A
Knowles	B
Kolberg	C
Korobago	C
Kranski	B
Kronen	B
LaFarge	B
*Lafont	B
Lamartine	C
Lamont	B
Langlade	B
Lapeer	B
Lawler	B
Lawson	C
Leelanau	A
Lena	A/D
Lenawee	B/D
Leola	B
LeRoy	B
Lindstrom	B
Lino	B
Linwood	A/D
Littleton	B
Lobo	D
Locke	B
Lomira	B
Longrie	B
Lorenzo	B
Lows	B/D
Loxley	A/D
Loyal	B
Ludington	B
Lunds	C
Lupton	A/D
Lutzke	B
Mackinac	B
Magnor	C
Mahalasville	B/D
Mahtomedi	A
Mahtowa	C/D
Manawa	C
Mancelona	A
Manistee	A
Manitowish	B
Mann	B/D
Maraglade	C
Marathon	B
Marcellon	C
Markeson	B
Markey	A/D
Markham	C
Marshan	B/D
Marshfield	B/D
Martinton	C
Matherton	B
Maumee	A/D
Mayville	B
McHenry	B
Meadland	C
Mecan	B
Medary	C
Meehan	B
Menahga	A
Menasha	D
Mendota	B
Menominee	A
Mequon	C
Meridian	B
Merrillan	C
Merwin	A/D
Metea	B
Miami	B
Michigamme	B
Mifflin	B
Milford	B/D
Military	B
Milladore	C
Millerville	A/D
Millington	B/D
Milton	C
Minocqua	B/D
Moberg	B
Monico	C
Montello	C
Montgomery	D
Mooselake	A/D
Moquah	B
Mora	C
Morley	C
Morocco	B
Mosel	C
Mosinee	B
Moundville	A
Mt. Carroll	B
Mundelein	B
Munising	B
Munuscong	B/D
Muscatine	B
Muskego	A/D
Mussey	B/D
Mylrea	C
Myrtle	B
Nahma	B/D
Namur	D
Navan	D
Nebago	C
Neda	C
Neenah	C
Nemadji	B
Nenno	C
Newaygo	B
New Glarus	B
Newson	A/D
Newton	A/D
New Vienna	
Nichols	B
Nickin	B
Norden	B
Norgo	D
Norrie	B
Northfield	D
*Norwalk	B
Nymore	A
Oakville	A
Ockley	B
Oconto	B
Oesterle	C
*Ogden	D
Ogle	B
Okee	B
Omega	A
Omena	B
Omro	C
Onamia	B
Onaway	B
Ontonagon	D
*Orienta	
Oshkosh	C
Oshtemo	B
*Ossco	
Ossian	B/D
Ostrander	B
Otter	B/D
Otterholt	B
Owosso	B
Ozaukee	C
Padus	B
Palms	A/D
Palsgrove	B
pardeeville	B
Parent	B/D
Pearl	B
Pecatonica	B
Peebles	C
Pelkie	A
Pella	B/D
Pence	B
Peshekee	D
Pickford	D
Pillot	B
Pinconning	B/D
*Pistakee	B
Plainbo	A
Plainfield	A
Plano	B
Pleine	D
Plover	C
Point	C
Port Byron	B
Poskin	C
Poy	D
Poygan	D
Prebish	C/D
Puchyan	B
Racine	B
Radford	B
Reedsburg	C
Renova	B
Rib	B/D
Ribhill	B
Richford	A
Richter	B
Richwood	B
Rietbrock	C
Rifle	A/D
Rimer	C
Ringwood	B
Ripon	B
Ritchey	D
Rockbridge	B
Rockers	C
Rockton	B
Rodman	A
*Rollin	D
Rondeau	A/D
Ronneby	C
Roscommon	A/D
Rosholt	B
Rotamer	B
*Rothschild	
Rousseau	A
Rowley	C
Rozellville	B
Rozetta	B
Rubicon	A
Rudyard	D
Ruse	D
Sable	B/D
St. Charles	B
Salter	B
Santiago	B
Sargeant	D
Sarona	B
Sartell	A
Sattre	B
Saugatuck	C
Saybrook	B
Saylesville	C
Sayner	A
Schapville	C
Scott Lake	B
Seaton	B
Sebewa	B/D
Seelyeville	A/D
Selkirk	C
Seward	B
Shawano	A
Sherry	B/D
Shiffer	C
Shiocton	C
Shullsburg	C
Sisson	B
Skanee	C
Skyberg	C
Soderville	A
Sogn	D
Solona	C
Spalding	D
Sparta	A
Spencer	B
Spinks	A
Spirit	C
Stambaugh	B
Stronghurst	B
*Suamico	
Summerville	D
Sundell	B
Superior	D
Sylvester	B
Symco	C
Symerton	B
Tacoosh	B/D
Tama	B
Tarr	A
Tawas	A/D
Tedrow	B
Tell	B
Terril	B
Thackery	B
Theresa	B
Tilleda	B
Toddville	B
Tonkey	B/D
Trempe	A
Trempealeau	B
Trenary	B
Troxel	B
Tula	C
Tuscola	B
Tustin	B
Udolpho	B/D
Underhill	B
Urne	B
Valton	C
Varna	C
Veedum	D
Vesper	D
Vilas	A
Virgil	B
Vlasaty	C
Wacousta	B/D
Wainola	B
Waiska	B
Wakefield	B
Wallkill	C/D
Warman	B/D
Warsaw	B
Wasepi	B
Washtenaw	C/D
Waskish	D
Watseka	B
Watton	C
Wauconda	B
*Waukechon	
Waukegan	B
Waupaca	B/D
*Wausau	
Wauseon	B/D
Wautoma	B/D
Waymor	B
Wega	B
Westville	B
Whalan	B
Wheatley	A/D
Whitehall	B
*Wien	D
Wildale	C
Will	B/D
Willette	A/D
Winnebago	B
Winneconne	C
Winneshiek	B
Winterfield	A/D
Withee	C
Worcester	C
Worthen	B
Wyeville	C
Wykoff	B
Wyocena	B
Yahara	C
Zimmerman	A
Zittau	C
Zurich	B
Zwingle	D

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Notes: Two hydrologic soil groups such as A/D indicates the drained/undrained situation.

* Denotes an inactive or dropped soil series.

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Table 2

Runoff Curve Numbers for Selected Agricultural, Suburban and Urban Land Use

Land Use in Watershed (%)		Land Use Description	CN's for Hydrologic Soil Group from Table 1: A	B	C	D		% Land use
	X	Cultivated Land[1] : without conservation treatment	72	81	88	91	=	
		: with conservation treatment	62	71	78	81		
		Pasture or range Land: poor condition, wetland	68	79	86	89		
		good condition	39	61	74	80		
		Meadow: good condition	30	58	71	78		
		Wood or Forest land: thin stand, poor cover, no mulch	45	66	77	83		
		good cover[2]	25	55	70	77		
		Open Space, lawns, parks, golf courses, cemeteries, etc. good condition: grass cover on 75% or more of the area	39	61	74	80		
		fair condition: grass cover on 50% to 75% of the area	49	69	79	84		
		Commercial and business areas (85% impervious)	89	92	94	95		
		Industrial districts (72% impervious)	81	88	91	93		
		Residential:[3] Average Lot Size — Average % Impervious[4]						
		1/8 acre or less — 65	77	85	90	92		
		1/4 acre — 38	61	75	83	87		
		1/3 acre — 30	57	72	81	86		
		1/2 acre — 25	54	70	80	85		
		1 acre — 20	51	68	79	84		
		Paved parking lots, roofs, driveways, etc.	98	98	98	98		
		Streets and roads: paved with curbs and storm sewers	98	98	98	98		
		gravel	76	85	89	91		
		dirt	72	82	87	89		

Add these numbers

Composite CN ______

[1] For a more detailed description of agricultural land use curve numbers refer to National Engineering Handbook, Section 4, Hydrology, Chapter 9, Aug. 1972.

[2] Good cover is protected from grazing, and litter and brush cover soil.

[3] Curve numbers are computed assuming the runoff from the house and driveway is directed towards the street with a minimum of roof water directed to lawns where additional infiltration could occur.

[4] The remaining permeable areas (lawn) are considered to be in pasture: good condition for these curve numbers.

TABLE 3. RAINFALL IN INCHES FOR A 2-YEAR 24-HOUR STORM EVENT FOR COUNTIES IN WISCONSIN.

County	Rainfall (Inches)	County	Rainfall (Inches)
Adams	2.8	Marathon	2.7
Ashland	2.6	Marinette	2.4
Barron	2.7	Marquette	2.7
Bayfield	2.6	Menominee	2.5
Brown	2.5	Milwaukee	2.7
Buffalo	2.9	Monroe	2.9
Burnett	2.7	Oconto	2.4
Calumet	2.5	Oneida	2.5
Chippewa	2.7	Outagamie	2.5
Clark	2.8	Ozaukee	2.6
Columbia	2.8	Pepin	2.8
Crawford	3.0	Pierce	2.8
Dane	2.9	Polk	2.7
Dodge	2.8	Portage	2.7
Door	2.4	Price	2.6
Douglas	2.6	Racine	2.7
Dunn	2.8	Richland	2.9
Eau Claie	2.8	Rock	2.9
Florence	2.4	Rusk	2.7
Fond du Lac	2.6	St. Croix	2.8
Forest	2.4	Sauk	2.9
Grant	3.0	Sawyer	2.6
Green	3.0	Shawano	2.5
Green Lake	2.7	Sheboygan	2.5
Iowa	3.0	Taylor	2.7
Iron	2.5	Trempealeau	2.9
Jackson	2.8	Vernon	3.0
Jefferson	2.8	Vilas	2.5
Juneau	2.8	Walworth	2.8
Kenosha	2.8	Washburn	2.7
Kewaunee	2.4	Washington	2.7
La Crosse	2.9	Waukesha	2.7
Lafayette	3.0	Waupaca	2.6
Langlade	2.5	Waushara	2.7
Lincoln	2.6	Winnebago	2.6
Manitowoc	2.4	Wood	2.7

TABLE 4. AVERAGE RUNOFF IN A 2 YEAR-24 HOUR STORM FOR SELECTED CURVE NUMBERS (in/acre).

Rainfall (Inches) From Table 3	Curve Number (CN) From Table 2[1]								
	60	65	70	75	80	85	90	96	98
2.0	0.06	0.14	0.24	0.38	0.56	0.80	1.09	1.48	1.77
2.1	0.08	0.17	0.28	0.43	0.63	0.88	1.18	1.58	1.87
2.2	0.10	0.20	0.33	0.49	0.69	0.95	1.27	1.67	1.97
2.3	0.13	0.24	0.37	0.54	0.76	1.03	1.35	1.77	2.07
2.4	0.15	0.27	0.42	0.60	0.82	1.10	1.44	1.86	2.17
2.5	0.17	0.30	0.46	0.65	0.89	1.18	1.53	1.96	2.27
2.6	0.20	0.34	0.51	0.71	0.96	1.26	1.62	2.06	2.37
2.7	0.23	0.38	0.56	0.77	1.03	1.34	1.71	2.16	2.47
2.8	0.27	0.43	0.62	0.84	1.11	1.43	1.80	2.25	2.58
2.9	0.30	0.47	0.67	0.90	1.18	1.51	1.89	2.35	2.68
3.0	0.33	0.51	0.72	0.96	1.25	1.59	1.98	2.45	2.78

(1) To obtain runoff depths for CN's and other rainfall amounts not shown in this table, use an arithmatic interpolation.

two curve numbers (Table 4)

(1.18 + 1.51) divided by 2 = 1.35 = 1.4" of runoff from the watershed

Computations

1.4" x 3200 acres divided by 12"/ft. = 373 acre-feet of water which is the total volume of runoff in a 2 year event

240 acre-feet divided by 373 acre-feet (wetland storage area) equals .64 or 64% of total volume of runoff will be contained in the wetland during a 2 year, 24 hour storm event

Proposed that 150 acres will be drained and developed:

240 acres minus 150 acres equals total available remaining storage after development, 90 acres or 90 acre-feet of water.

90 acre-feet (wetland storage volume) divided by 373 acre-feet (total volume of runoff in watershed during 2 year, 24 hour storm event equals 0.24 or 24% of total volume of runoff will be contained in 90 acre wetland during a 2 year, 24 hour rainfall event.

64% (240 acres of wetland storage capacity of watershed runoff during a 2 year, 24 hour storm event) minus 24% (90 acre wetland storage capacity) of watershed runoff during 2 year, 24 your storm event equals a 40% loss of storage capacity for runoff in the watershed during a 2 year, 24 hour storm event if 150 acres of wetland are filled. Or, the functional significance of the 240 acre wetland to store floodwater will sustain a 62% reduction in wetland storage if 150 acres of wetland are filled. (This is determined by dividing 150 acres of wetland filled by 240 acres of original wetland).

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The Charles River Watershed: A Dual Approach to FloodPlain Management

Arthur F. Doyle
U.S. Army Corps of Engineers

INTRODUCTION

The Charles River Watershed is only about 15 miles wide and 31 miles long, yet the river meanders over 80 miles from its headwaters at Hopkinton, Massachusetts to its terminus in Boston Harbor. Along its path, the river's surroundings change dramatically.

The Charles River originates at picturesque Echo Lake Reservoir. Traveling downstream from the lake, one sees the marshes and swamps that nurture and sustain native fish and wildlife. Then, about halfway through the journey, the river banks change to a more suburban scene of scattered homes and light industry. Getting closer to the river's end, the scene shifts to apartment buildings, parking lots and shopping complexes. Finally, the river empties into commercialized Boston Harbor. All of Boston's wetlands were long ago filled and paved over for high-rise offices, universities and parkways. On the grassy strips between the river and parkways, one can usually see bike riders, joggers and picnickers.

Just as the river's scenery changes, so does its hydrology. These hydrological differences, in effect, divide the watershed into two separate systems. In the lower urbanized section, storm runoff races across the paved surfaces straight into the river quickly raising water levels and accelerating flows. In the less developed middle and upper sections, however marshes and swamps soak up water and storm runoff is sluggish.

These differences become apparent when storms sweep through the area. In 1955, Hurricane Diane hit the watershed and caused flood damages in excess of five million dollars in the lower portion. Boston and Cambridge suffered the most, while damages in the middle and upper watershed were minimal.

The record flooding that followed Hurricane Diane focused new attention on the Charles. In 1965, Congress directed the Corps of Engineers to study ways to control flooding in the urbanized lower basin and prevent similar damages from occurring in the rest of the watershed. During the study, the Corps developed two different solutions to comply with Congress's request. It is these two approaches to flood plain management that make the Charles River Project unique.

It seemed obvious that corrective measures would have to be taken in the lower watershed. Apart from urbanization, another problem was caused by an old dam that had been built in 1910 to fill in unsightly tidal marshes and putrid swamps. This dam, located beneath the Museum of Science, impounds what is known locally as the "Basin." The dam was designed to maintain the water at a constant level of 2.48 mean sea level (msl). When heavy rainstorms were combined with tides as high as six feet msl, the dam's sluice gates were unable to pass the excess flows. The water had no place to go except up--thus flooding occurred.

The Corps and consultants found that it was impossible to improve the old dam for flood control. Therefore, in 1968 the Corps recommended that a new dam be built farther downstream. The project was approved by Congress and construction started in February 1974. The new dam was dedicated in May 1978, and was recently turned over to the Metropolitan District Commission (MDC) for operation and maintenance.

The major feature of the dam is its large pumping station that contains six vertical lift pumps, each capable if discharging 630,000 gallons of water per minute. Three navigational locks have been constructed, two for recreational craft and the third for commercial ships. The first two locks measure 200 feet long and 22 feet wide, and the larger lock is 300 feet long and 40 feet wide. Overhead is an enclosed walkway that connects free-standing stair towers and the pump station with the navigation locks. The control station for operating the locks is also located here. At the southern terminus of this walkway is a police boat facility, which was added to the project at the request of the MDC. It is at an ideal site that offers ready access to both the river and harbor.

Another feature of the dam is its fishway. Designed by the Corps and the U.S. Fish and Wildlife Service, it allows fish to pass from Boston Harbor to the river whether the tide is higher or lower than the "Basin."

Sightseers are able to view the fishway, dam and locks from a public walkway that extends across the project and on into a small park.

NATURAL VALLEY STORAGE

As the study in the lower basin was coming to an end, the Corps was given a perfect opportunity to observe the middle and upper watershed in

action. In 1968 a big storm hit the watershed. A Corps reconnaissance team began commuting to the wetlands to observe them in action. The 1968 flood crest moved so slowly that the team was able to measure the crest one day, go home for the night, then come back the next morning to pick it up slightly below the point where they had left it. While runoff from the 58-square mile lower basin crested at the old dam in a few hours, the upstream peak took 40 days and the entire volume of storm water took a month to reach the structure. One stretch of the river was widened from 50 feet to nearly a mile. The wetlands had acted like a reservoir, first holding back floodwaters and then gradually releasing them.

The importance of the wetlands in the middle and upper watershed is reflected by the flood hydrographs recorded at the USGS gaging station at Charles River Village. Maximum floodflows in terms of cubic feet per second (cfs) are extremely low in the Charles River when compared to the adjacent Blackstone River, which has relatively few wetlands. In the 1955 flood of record, which was almost duplicated by the 1968 flood, the peak discharge at Charles River Village was only 3,220 cfs, while the peak discharge on the Blackstone River at Northbridge was 16,900 cfs. Following this peak discharge on the Charles, about 50,000 acre-feet of storm water slowly flowed through the gaging station over the next month. This is equivalent to about five inches of runoff from the Charles' drainage area of 184 square miles. For all practical purposes, the wetlands proved to have the same storage capacity (50,000 acre-feet) as the North Springfield or Birch Hill flood control reservoirs.

These measurements confirmed the value of the basin's wetlands as a natural flood control system, but the Corps study team realized that this natural hydraulic system could soon disappear. In the late 1960's substantial development threatened wetlands in the mid and upper watershed.

Boston's second circumferential highway, I-495, was already under construction, which lent a sense of urgency to the study team's work. This highway would open up the mostly rural countryside to accelerating growth. As pavement and drainage systems replaced vegetation, storm runoff would become more intense and the land's storage capacity would decrease. This unchecked growth would increase upstream flood damages and threaten the safety of downstream residents.

The following hydrologic chart shows a mathematical determination of what might happen at the Charles River Village gage if the watershed's storage capacity decreases:

Loss in Water Storage (acre-feet)	Discharge at Charles River Village (cfs)
0 (1955 and 1968 floods)	3,200
11,000	4,500
20,000	6,000
32,000	9,000
39,000	12,000
44,000	15,000

Understanding the potential danger facing the wetlands, the Corps study team decided that the surest way to "save" the wetlands was to buy them.

There are approximately 20,000 acres of wetlands in the watershed. It would be impossible to preserve all these swamps and marshes, as this would be too restrictive for an orderly development of the basin. The late Elliot Childs, Chief of the Corps Hydrology Branch at that time, determined from his analyses that preserving the most critical acres would prevent serious increases in future flood stages. Childs found that approximately 10,000 acres, about half of the total wetlands, had major floodwater retention capabilities. Geographically, these acres are separate, widely distributed units. Hydrologically, however, they act as a single integrated system.

After studying the 10,000 acres in detail, about 8,500 acres were selected for acquisition. Located in 17 natural storage areas, their sizes range from 118 to 2,340 acres. Parcels smaller than 100 acres were not considered because of their low storage capacity. Only three areas are adjacent to the Charles; the others are located in tributary sub-basins.

A land-buying program of this magnitude would need public support. Communication lines established back in 1966 were kept open to the 16 cities and towns located in the affected areas. A Citizens Advisory Committee (CAC), consisting of more than 30 representatives of local communities and businesses, acted as a liaison with the general public. They passed on to the Corps the wishes of interested persons and organizations and served as a sounding board for study proposals. At the conclusion of the study in 1971, the Corps sponsored informal information meetings and a formal public meeting. More than 600 persons attended. They unanimously endorsed the acquisition program. Members of the CAC then began visiting and writing to State and Federal officials and legislators. Soon the Massachusetts Legislature became supporters of the program.

The time was right for this type of program because flood damages were climbing each year and environmental groups were looking for new ways to control flooding. If no action was taken, it was estimated that in 30 to 40 years, 100 million dollars would be required for flood control

structures. The estimated 10 million dollars needed for acquiring and preserving the land seemed like a bargain. For the first time in its history, Congress, in March 1974, authorized and funded money to buy land for nonstructural flood control. What followed is the program called Natural Valley Storage (NVS).

Between 1974 and 1977 the specifics of the program were worked out and the wheels were set into motion. Hydrologists measured storage capacities and inflow/outflow characteristics of each NVS area, engineers determined boundaries, and real estate specialists researched titles, appraised land and negotiated for property. The New England Division purchased the first acreage in May 1977. The acres are being acquired according to priorities that consider location, volume of storage capacity and degree of development threat. The acquisition phase was completed in 1984. In accordance with the law, the Corps is paying fair market value for the wetlands.

Some property owners were unwilling to sell their land even though they support NVS. In other cases, critical wetlands were owned by land trusts such as the Massachusetts Audubon Society. A special restrictive easement was prepared for these situations. It allowed the owner to retain title to the property, but gave the government access for inspections to ensure the land is preserved in its natural state.

Naturally, there were questions. Many were worried about the tax revenues that their communities would lose when privately owned wetlands were taken off the tax roles. One town for example, had 23 percent of its land designated as natural valley storage. To compensate for tax losses, the Massachusetts Legislature authorized reimbursements to the affected cities and towns. Therefore -- nobody lost.

Once the land was in the public domain, the Corps retained its ownership but not its management. The Massachusetts Fisheries and Wildlife Division manages the area through a lease arrangement with the Corps; they enforce laws, stock fish and improve wildlife habitats.

Although it was the first and only project of its kind, NVS can be duplicated elsewhere. In fact, thousands of inquiries have been received requesting information about the program. But, for a natural valley storage project to be feasible, these three conditions must exist:

1. The storage areas exist in a natural state;
2. Little or no flood damages exist, but the potential is there; and
3. Loss of the areas is imminent.

In the right circumstances, a nonstructural, environmentally-oriented prevention project like this can be an effective and economical flood management choice--an alliance with nature of a confrontation.

SUMMARY

In January 1979, the Charles River once again experienced near record flooding. The new dam's pumping station successfully expelled the Charles River's floodwaters from the "Basin" preventing an estimated 14 million dollars in damages. And the wetlands once more performed effectively, first storing floodwaters and then gradually and safely releasing them. The storage capabilities of the natural valley storage areas were also effectively utilized in June 1982. These floods demonstrate that this dual approach to flood plain management is not only unique--it works.

Predicting the Impact of Vegetation on Storm Surges

Christopher D. Miller
Greenhorne & O'Mara, Inc.

INTRODUCTION

The generation of storm surge on the open ocean/continental shelf and its transformation in shallow coastal waters has been studied in some detail (see, for example, Murty, 1984). Less is known about the same phenomenon in estuarine areas that may be characterized by either limited or extensive exposure to the open coast/open ocean.

The influence of vegetation is obviously related to the types and areal extent of vegetation present within a particular estuary. Figure 1 shows a typical cross-section perpendicular to the coastline. Because of the tidal influence the location of species like Spartina alterniflora proliferate near the coast whereas less salt-tolerant species such as Juncus roemerianus, as well as shrubs and trees, are found either farther inland or on higher ground. The salt-tolerant species interact with tidal waters; the more sensitive species may only interact with storm-generated waters.

Within U.S. East Coast and Gulf Coast estuaries the fraction of the total surface area represented by marsh is indicative of the potential hydrodynamic influence of this vegetation. The vegetation must be of fairly broad extent to have a marked effect. Although a patch of marsh grass with an area of several thousand square feet can impede local flow, it will not noticeably alter storm flood elevations. The following table shows that in the majority of East and Gulf coast cases sufficient marsh exists to anticipate a significant impact.

Because of the complexity of land features and watercourses within an estuary, the pattern of storm elevations is two-dimensional, i.e., a uniform one-dimensional decay (or increase) of the storm surge with distance inland is the exception rather than the rule. Figure 2 shows the distribution of storm surge elevations for Louisiana caused by Hurricane Audrey. Prominent characteristics are the lower flood levels in the western marshes relative to the open coast surge and the relatively higher elevations along portions of the Mississippi River.

SOME EARLY LABORATORY EXPERIMENTS

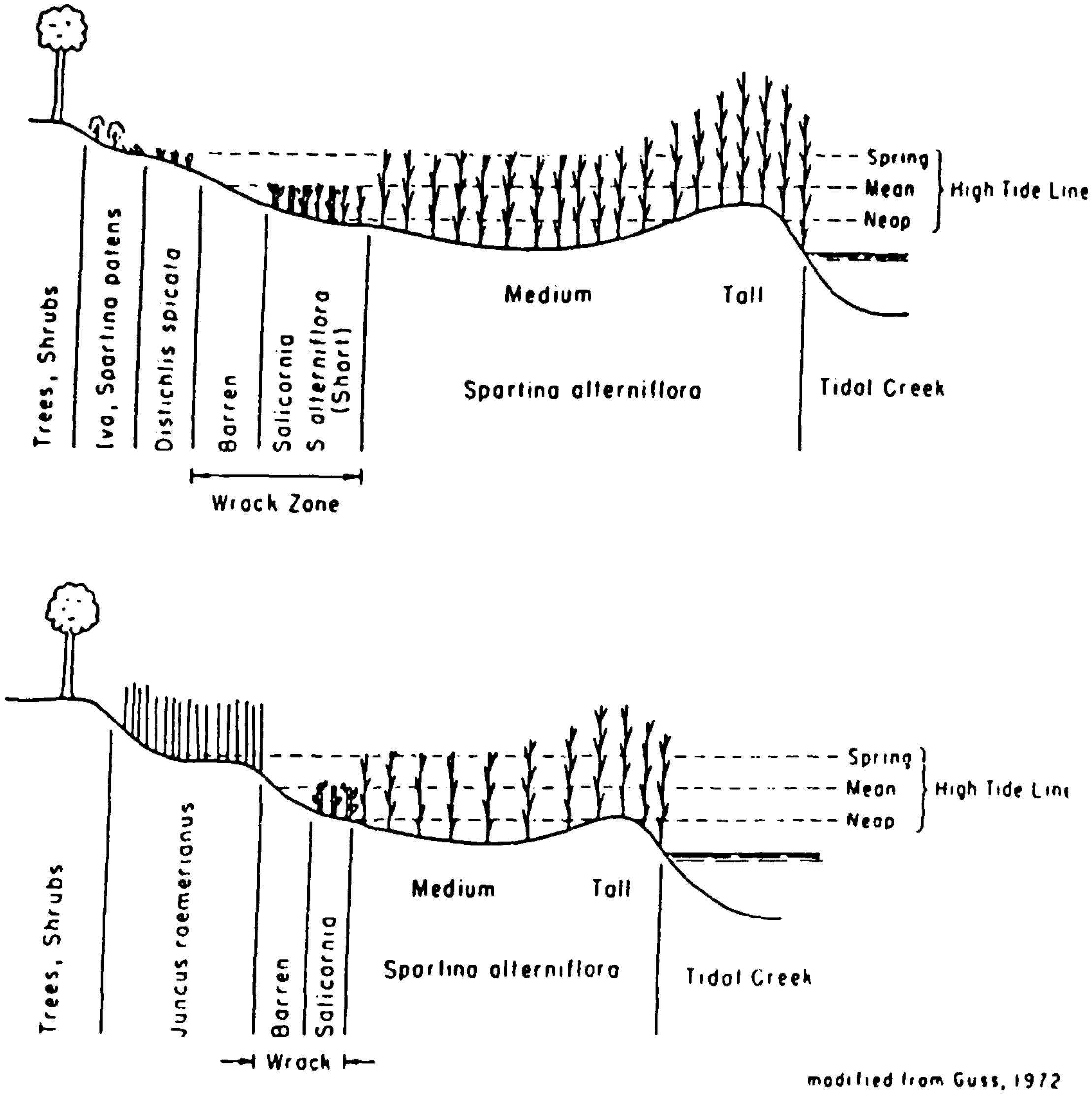
Tickner (1957) conducted laboratory simulations of the effects of vegetation on wind setup (surge) and waves. Vegetation was represented by wire mesh screening placed across the bottom of an enclosed channel. The spacing between the roughness elements (screens) was varied from 0.2 ft. to 0.8 ft. A blower within the tank created the wind conditions for surge and wave generation. At the downwind end of the tank the water surface intersected a 1 on 10 sloping beach (ramp). The tank width was 1 foot; thus, one-dimensional flow in the horizontal plane (parallel to the wind direction) was implicitly imposed.

Within the relatively shallow tank the depth of the vegetation (ratio of height of roughness element to water depth) varied from approximately 0.3 to 2.0. Therefore, both submerged and exposed vegetation cases were considered. Through several runs the relationship

Table 1

REGION (# CASES)	AVG. BAY WIDTH (FT)	% MARSH IN BAY	AVG. MARSH WIDTH (FT)
Long Island (6)	11,540	25.5	2,943
New Jersey (6)	13,384	62.3	8,338
Delmarva(5)	20,233	52.4	10,602
Outer Banks (6)	57,675	15.8	9,113
Southern North Carolina (5)	3,110	44.0	1,368
East Florida (13)	3,979	18.9	752
Texas (8)	33,150	29.4	9,746

Data from "Geometry of Inlets on the Atlantic and Gulf Coasts of the United States, Galvin et al, October, 1973.



COMPOSITE COASTAL VEGETATION ZONES
FROM SOUTH CAROLINA

FIGURE 1

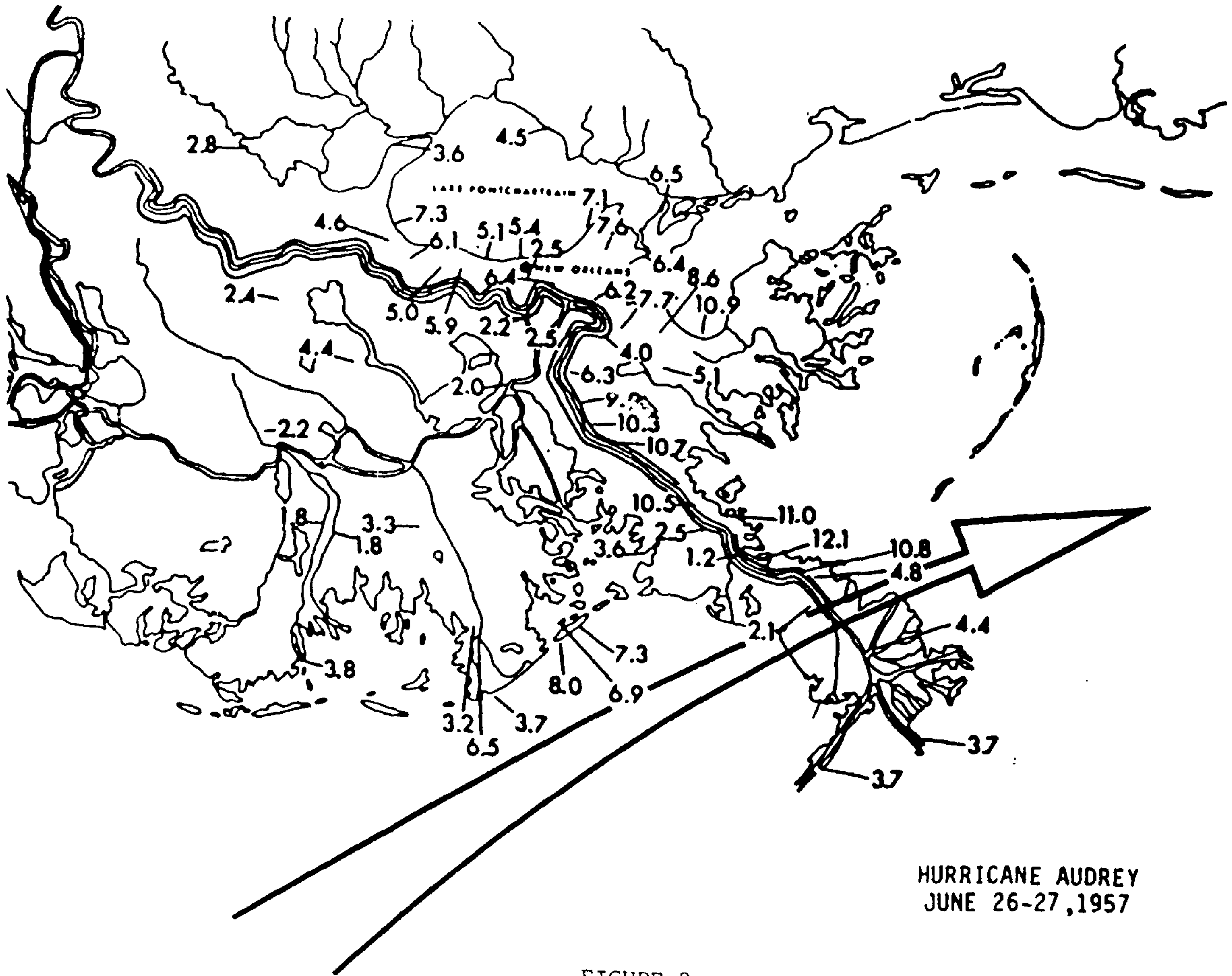


FIGURE 2
(from Harris, 1963)

between Manning's n and the spacing of the roughness elements was established. Not unexpectedly, the Manning's n increased with decreased spacing and with decreasing water depth. When the water depth was only half the height of the vegetation, there was a remarkable decrease in the wind setup, i.e., the wind sheltering effect of the protruding vegetation reduced the wind stress on the water surface and produced a setup that was 8% of the setup obtained with a smooth (unvegetated) bottom. In contrast, with the roughness elements submerged, the setup was as much as twice the smooth bottom value. In the latter case the retarding action of the vegetation, combined with the upward slope of the beach ramp, apparently prevented the water that was driven toward the beach from being released, thereby creating a high, local setup. This seemingly counter-intuitive result will be discussed later in the context of two-dimensional horizontal flow simulations.

VARIOUS FORMS OF FRICTIONAL RESISTANCE

Water moving through an estuary is affected by wind stress at the water surface, frictional stress at the bottom, and form drag due to obstructions in the water column. There are other influences, e.g., gravitational pressure gradient due to tilt of the water surface, the Coriolis effect, lateral eddy viscosity, etc., but within the context of the present discussion on vegetation effects only limited reference to these influences is necessary. Detailed treatment of the factors that determine wind stress is given in other papers, e.g., Large and Pond (1981). Bottom stress is usually parameterized in terms of the Darcy-Weisbach formula or the Manning's formula. For two-dimensional flow the bed shear stress τ_b according to Darcy-Weisbach is

$$\tau_b = \frac{\rho f|U|(u,v)}{2}$$

where ρ = density of water, f = the friction factor, $|U| = (u^2 + v^2)^{1/2}$, the magnitude of the velocity vector in which u and v are the two instantaneous velocity components.

In terms of Manning's formula Eq. (1) becomes (in English units)

$$\tau_b = \frac{\rho g n^2|U|(u,v)}{(1.49)^2 d^{1/3}}$$

where g = gravitational constant, d = water depth and n = Manning's coefficient

A comparison of these two equations indicates that the friction factor should be a function of water depth and some data exist to define this dependence (e.g., FEMA, 1987).

When obstructions are present in the water column, they induce drag on the moving water. It is convenient to conceptualize a wooded area as a stand of evenly spaced cylinders. The total resistence is, then, the summation of the contribution from each cylinder. The drag force can be re-cast as a stress (force normalized by the flow area) as follows:

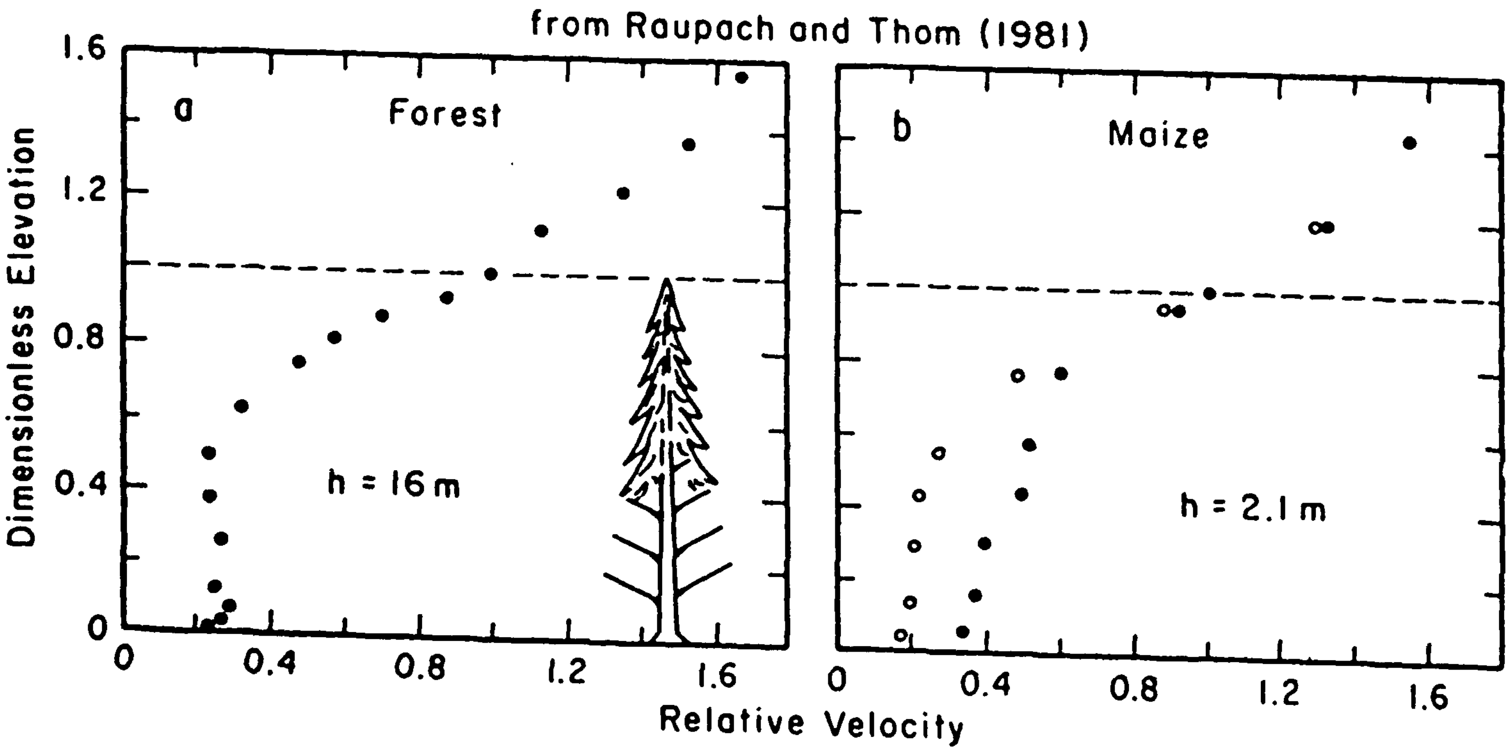
$$b = \frac{\rho C_D(NDh)|U|(u,v)}{2}$$

where C_D = dimensionless drag coefficient, N = number of cylinders per unit flow area, D = cylinder diameter, h = submerged height of the cylinders.

Because each of the above expressions is similar in form, i.e., proportional to the square of the velocity, they are somewhat interchangeable. For example, an "effective" Manning's n can be defined which includes the normal "bottom stress" Manning's n and a "form drag" Manning's n (via Eq. 3). Thus, a common denominator can be established to relate the resistance offered by ground, marsh, trees, etc. Appropriate friction factors can be obtained from various sources (e.g., FEMA, 1987; Christenson and Walton, 1980; Wang and Christensen, 1986).

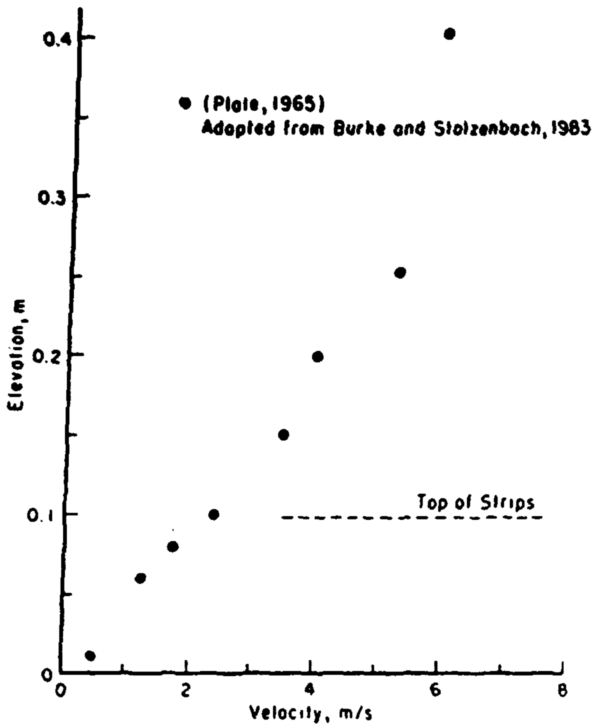
EFFECT OF VEGETATION ON WIND AND CURRENT VELOCITY PROFILES

The stress, and, therefore, the momentum, that wind imparts to the water surface is a function of surface roughness and the vertical profile of the horizontal wind velocity. Surface roughness is controlled by the presence of surface waves and by any objects that protrude through the water surface. The wind profile is controlled partially by the presence of roughness elements at the water-air interface, e.g., vegetation, buildings, topographic features. Surface roughness and wind profiles are obviously interrelated. Experiments have demonstrated that vegetative obstructions appreciably alter the normal velocity profile in the air and water columns. Figure 3 shows the effect of a wooded area and maize, independently, on the vertical distribution of the horizontal wind velocity. There is a relatively sharp break, from high velocity to low velocity, in the vicinity of the crest of the canopy. For submerged marsh grass laboratory tests with flexible, plastic strips show a similar pattern of diminished velocity with depth in the water column (Figure 4). If the vegetation protrudes through the water surface, then a composite of the two cases applies, i.e., a parabolic-like wind profile in the air column combined with a similarly-shaped current velocity profile in the water column. The vegetation acts in a dual mode to reduce wind stress and wind-generated currents. This scenario has been formally modeled by Whitaker et al. (1973) for the situation of a hurricane moving



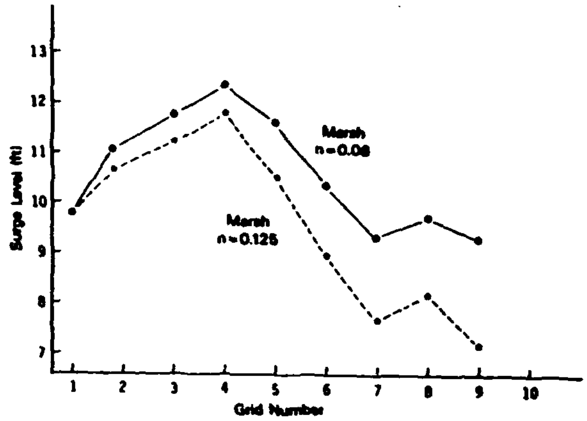
VELOCITY DISTRIBUTION IN TWO VEGETATIVE CANOPIES

FIGURE 3



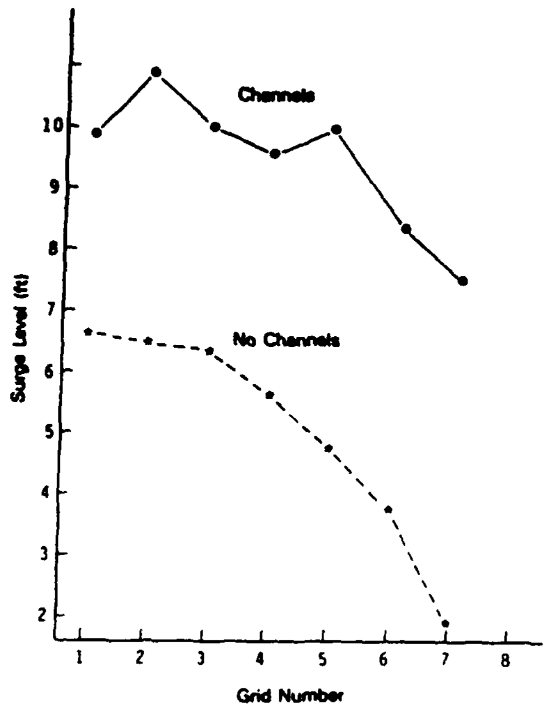
VELOCITY DISTRIBUTION IN FLEXIBLE STRIP CANOPY

Figure 4



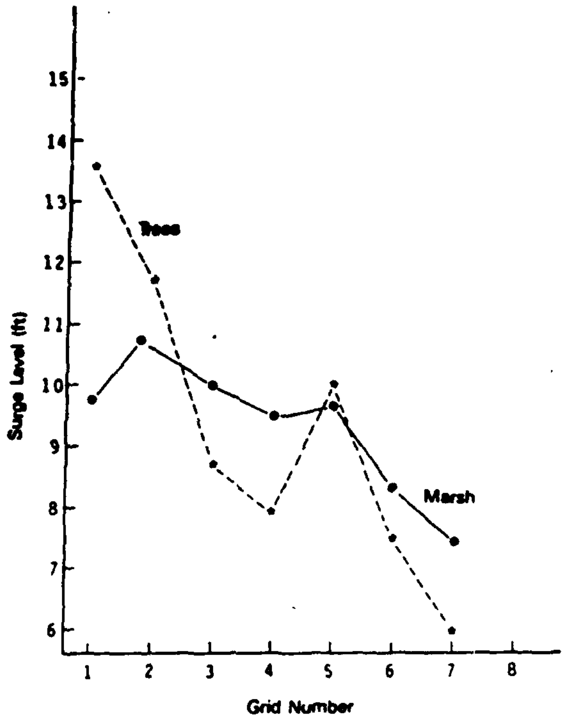
INLAND SURGE PROFILE

Figure 5



INLAND SURGE PROFILE

Figure 6



INLAND SURGE PROFILE

Figure 7

across Lake Okeechobee. This lake has a large and vigorous marsh stand. In their model the canopy is composed of rigid cylinders evenly spaced over the bottom. Both the wind and current sheltering aspects of the canopy are accounted for with simple parameterizations. Using their formula it can be shown that less than 5% of the wind shear stress above the vegetation reaches the water surface when a modest canopy is present, e.g., a stem diameter of 0.1 foot, a spacing of one roughness element per square foot, and a height above the water surface of 1 foot or more. For non-rigid vegetation a larger percentage of the wind stress penetrates to the surface.

NUMERICAL EXPERIMENTS

For the case of the effect of the the well-mixed estuary on storm conditions the implementation of a two-dimensional long-wave numerical model is adequate to describe the mean flows and water elevations. Several models exist that fulfill this role. The storm surge model developed by the Federal Emergency Management Agency (1987) is a versatile tool in that: (1) it tracks the moving water-land interface as flood waters advance and recede over lowlands; (2) physical features with dimensions smaller than the grid size can be resolved, i.e., subgrid features such as barriers and channels are accounted for; (3) the effect of various types of land cover on modifying the air flow and the water flow are taken into account. This model has been exercised for a typical estuary to test its response to different storm conditions and estuary features (see Miller and Wei, 1987). These sensitivity tests included some investigation of the role of vegetation. A few cases are described here. In all cases a severe, "100-year" hurricane was modeled.

The "test" estuary is characterized by barrier islands along the open coast, landward of which there are tidal lowlands, small and large embayments, obstructions of various types, and rivers and canals. The cases to be discussed here focus on the rivers and tidal marshes. Within the two-dimensional grid there is a sub-area that contains a large, meandering river moving through a tidal marsh. Flood elevations along a line drawn perpendicular to the coastline and passing through this sub-area are used to describe the model response to varying conditions. Figures 5, 6, and 7 show the surge elevation profile starting near the coastline (grid number 1) and terminating at an inland location. Each grid number represents a distance increment of one nautical mile, e.g., grid number 5 is 5 nautical miles from the open coast.

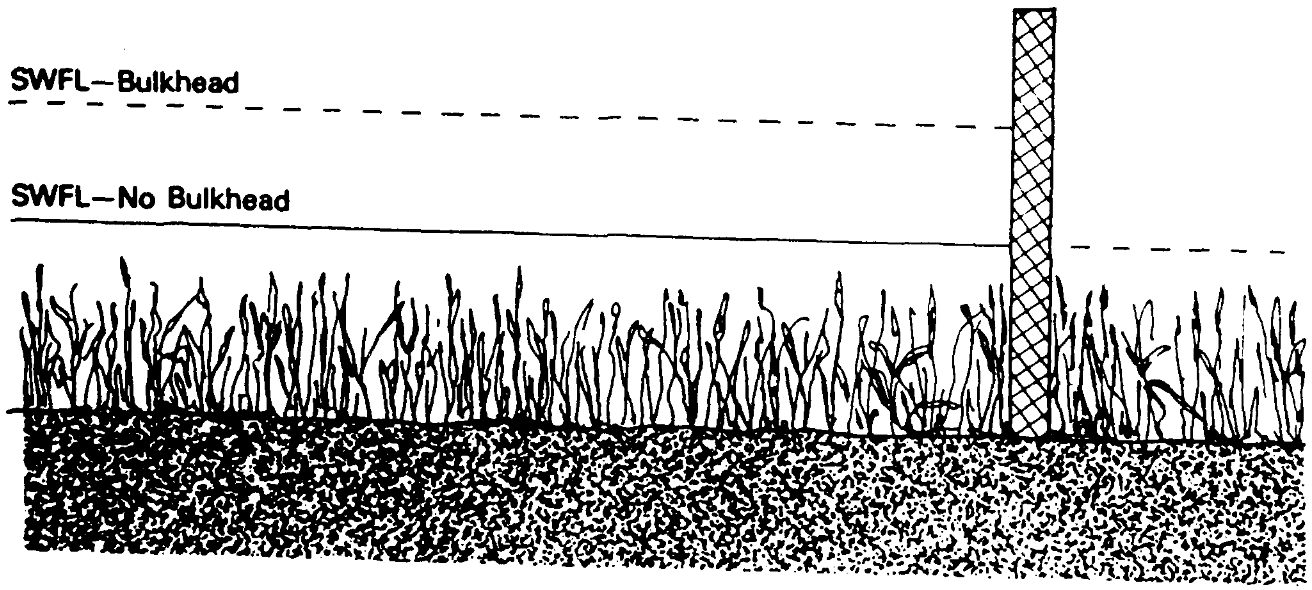
In Figure 5 the Manning's n associated with the marsh grass is varied. The value n = 0.06 could be associated with a marsh grass like small Spartina alterniflora and n = 0.125 with Juncus roemerianus. The more dissipative marsh grass attenuates the surge more rapidly such that the difference between the two curves increases with distance inland. The surge does not decay monotonically due to the competing effect of the river which conveys open coast water upstream (upland) very efficiently. Figure 6 demonstrates what happens when this channel is removed and only marsh grass is specified near the open coast. Starting with a reduced elevation, there is a steady decrease in the surge with distance inland. The respective final upstream elevations are radically different. This result has implications for channelization projects.

In Figure 7 the river is re-inserted but the marsh is replaced totally by forest. The effective Manning's n for the trees is more than twice the value for the marsh grass. At the coast the computed surge is actually higher for the trees than for the marsh grass. This is qualitatively similar to the effect observed in Tickner's experiments. The substantial resistance to flow offered by the trees leads to a piling up of surge water near the coast. This water is retained locally by the trees so that the accompanying flood levels farther inland drop more precipitously for the trees than for the marsh. The marsh tends to diffuse the flood waters more uniformly over the grid. As tree density increases (Manning's n increases) the trees near the coast act more like a solid wall, i.e., increasing near-coast flood levels but decreasing levels inland.

In another experiment a high structure at the upland end of a large embayment was inserted, replacing low-lying marsh (see Figure 8). The surge level increased by 0.08 foot. In other words, a buffer of vegetation can mitigate flood levels locally by providing storage area and not presenting a solid barrier to the flow.

CONCLUSIONS

Methods are available for resolving the effects of vegetation on tidal and storm waters. In general, the presence of vegetation mitigates flood levels and flood velocities, although in some cases local flood elevations may actually increase due to the large resistance offered by dense, rigid vegetation. The modeling option can be used to assess some of these effects but, to date, there has been limited reliance on this approach.



STILLWATER FLOOD LEVEL (SWFL)
WITH AND WITHOUT BULKHEAD

FIGURE 8

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Wetlands and Flooding: Assessing Hydrologic Functions

Larry A. Larson
Wisconsin Department of Natural Resources

INTRODUCTION

The hydrologic functions of wetlands are not evaluated as often or as accurately as they should be. Reasons include 1) wetland evaluations are often done by non-engineers (biologists, botanists, etc.) who do not understand hydrologic processes or 2) engineers have not made available the necessary data or methods so that non-engineers can make such assessments.

This paper will make three general points:

1. Data and research is needed, in a general sense, determining the importance of wetlands to flood storage.
2. The methods needed to assess the importance of a particular wetland to flood storage or flood conveyance are readily available, and floodplain engineers can readily perform these calculations. This paper will talk about some of the "how."
3. Regulations that are designed to preserve floodplains can be used to protect and preserve wetlands; such regulations can serve dual purposes.

DATA AND RESEARCH NEEDS

Very little has been done on a national basis to accumulate data to indicate the impacts of wetlands or wetland losses on flooding. Richard Novitzki talked about his data in Wisconsin, New York and a few other regions. I interpret that data to show the following:

1. The impact on flooding will be to reduce peak flood discharge if wetlands are present, but it may attenuate or extend the duration of flooding. Duration is extended because water is stored in the wetland while the flood peak passes but then the water flows back into the river and passes downstream extending the period of flooding. Wetland losses, therefore, can result in loss of flood storage and can increase downstream flood profiles and downstream flooding.
2. The greater the ratio of wetlands to the total watershed area, the greater the reduction in flooding.
3. In general, wetlands have greater impacts on small floods. That stands to reason, since wetlands may store nearly the same volume of water in the 2-year flood as in the 100-year flood. That storage will represent a greater percentage of the total volume of the smaller flood--hence, the greater the impact.
4. The greatest proportional impact on the flood peaks occurs in the 0-10% ratio of wetlands/watershed area. Larger ratios (up to 40 percent were studied) continue to decrease flood peaks but at a decreasing rate.

This data and what it demonstrates are important in overall assessment of wetlands for planning or policy-making purposes. However, additional data is needed for more specific assessment. Assessors should, at the minimum, be able to readily identify those wetlands that may have a significant impact on flooding. Some potentially useful parameters are listed below:

a. The type of river basin;

b. The type of wetland;

c. Basin size; and

d. Percentage of wetlands or certain type of wetlands in the floodplain and in the floodway (The importance of that will be explained below.).

Unanswered questions include:

1. Can much of the above data be regionalized?
2. Would a computerized data base, some kind of geographic information system containing data on both floodplains and wetlands, provide a source to quickly identify key factors which could then be generalized to identify wetlands that might have a significant impact on flood storage?

METHODOLOGIES FOR CALCULATING HYDROLOGIC IMPACTS ON WETLANDS

A methodology exists to determine the impact of a specific wetland on flood peaks. Floodplain engineers frequently use flood storage models

using the basic computer models developed by the Hydrologic Engineering Center (HEC) for the U.S. Army Corps of Engineers in Davis, California. A number of these techniques include HEC-1 and HEC-2 to determine the basic floodplain, the floodway and the floodfringe. Floodplains are those areas covered by water during the one percent flood. Floodways are the river and the immediate overbank areas necessary to convey flowing water and discharge the flood without an increase in upstream flood elevation. They often include wetlands. Floodfringe areas are those areas that commonly store water while the flood passes.

Floodfringe areas are the outer edges of the floodplain where the water is not moving, but is standing. Wetlands are also commonly included in the fringe. Storage models look primarily at the impact of any storage area on the discharge and downstream elevation of a flood. The model uses topography to calculate storage volume and impact. To the model, it is irrelevant whether that storage area is a wetland. If you believe that a particular wetland has an impact, your state floodplain management people, the Corps of Engineers or other engineers can calculate the impact.

This discussion, of course, focuses on riverine and not on coastal wetlands. The impact of coastal wetlands in terms of reducing flood elevations boils down to their impact on reducing the wave heights and wave rump, i.e., the inland run of the wave. Vegetation must apparently be quite thick in order to substantially reduce flood heights in a major storm. Not much research or data is available to help calculate this function.

USING FLOODPLAIN REGULATIONS TO PROTECT WETLANDS

Let's review floodplain regulation to determine how they might protect wetlands. Minimum standards incorporated in most floodplain regulations are:

- **Floodfringe areas.** Pursuant to most state and local regulations, floodfringe areas can be developed as long as any new structure in the fringe area is placed on fill and properly elevated above flood levels.

- **Floodway areas.** Floodway standards generally prohibit fill since any fill in this area would obstruct flows and increase flood levels upstream. Some experts have estimated that 80-90 percent of all wetlands are in the floodplain, and data may show that a good share of them are in the floodway. If this is true, support of and enforcement of floodplain regulations will help protect critical wetlands.

Many local communities have expanded floodplain standards which go beyond national or state minimums. These provide even more effective wetland protection. Many communities use open-space standards for the entire floodplain. Such standards allow only open-space uses and prohibit filling and building of structures in the entire floodplain. This works best in communities where there is little or no existing development in flood hazard areas or where there may be ample alternative development sites available. Communities should be encouraged to use that approach wherever possible.

In Wisconsin, cities and villages are now required to regulate both floodplains and wetlands. We have developed model ordinance language so that these cities and villages can add wetland protection to their floodplain ordinances or vice versa.

SUMMARY

In summary, it is important that wetland and floodplain managers work closely to develop a data base and to identify critical flood storage areas that include wetlands. Additional regional studies of wetland flood storage relationships are needed. Management experience would indicate that a method is needed to quickly "red flag" potential storage areas that may be most significant in reducing flooding. Once those wetlands are identified, a more detailed study can be done using existing methodology. Methods of arriving at that initial flagging decision would include the calculation of the percent of the watershed taken up by the wetlands. For example, a rule-of-thumb used in the State of New Hampshire is that if the wetland/watershed ratio is one percent or greater, wetlands may have significant flood storage value and deserve more detailed analysis. If the ratio is less than that, wetlands are not seriously considered for flood storage purposes.

The wetland/watershed ratio seems to be a promising quick method for looking at flood storage significance. As pointed out in the Charles River Study, the first 10 percent of the total watershed area in wetlands seems to be most important. For the Charles River, it was calculated that those wetlands comprising five percent of the total watershed would be sufficient to maintain the existing flow regime without significantly increasing downstream flood elevations.

Wetland managers need to work closely with local communities to encourage them to consider wetland and floodplain regulations as a package. Floodplain studies that have been done throughout the nation include over $650 million worth of studies by the Federal Emergency Management Agency. This kind of data needs to be better utilized.

Cumulative Impacts of Agricultural Land Drainage on Watershed Hydrology

Michal J. Bardecki
Ryerson Polytechnical Institute

INTRODUCTION

Heavy soils, low gradients and reasonably abundant precipitation in the province of Ontario somewhat limit the full agricultural potential of this area. As a result, in the southern part of the province where geological and climatic factors are most amenable to agriculture, land drainage has been extensive.

Agricultural land drainage most commonly involves the installation of buried tile drains to convey water from the surface down into the soil. In order to permit egress of this water, outlet drains consisting of engineered ditches and channelized streams are constructed.

Drainage provides three major benefits:

1. it increases the level of fertility due to increased aeration, higher soil temperature, improved soil structure, etc.;
2. it increases yields due to earlier germination, a longer growing season, and a higher soil fertility; normally, a greater choice is available regarding crop selection; and
3. it permits greater efficiency of farm operations (for example, soil bearing strength is increased allowing for greater mechanization).

Nonetheless, significant environmental effects result from drainage (Hill, 1975). Among these is the adverse impact on wetland areas. It has been estimated that 85 percent of wetland area lost in southern Ontario is attributable to agricultural land drainage (Bardecki, 1981).

Although drainage of agricultural lands started as early as the 1840s in Ontario (Kelly, 1975), the practice has shown great expansion in recent decades. This has meant increasing intensity of drainage in traditional areas (especially in the extreme southwestern and far eastern portions of Ontario), and also substantial expansion of drainage into new areas (Bardecki, 1984). Concern has been expressed that the loss of the wetlands due to increasing agricultural drainage has led and would lead to deterioration of the hydrological characteristics of streams in southern Ontario.

The often accepted wisdom, attributed to Alexander von Humboldt (Coode et al., 1977), is that wetland areas act as sponges (that is, they provide for storage of water during periods of high flow and release the water gradually, augmenting the low periods in streamflow). Few publications advocating wetland protection do not include this as a key value of wetland areas.

METHOD

Eight river basins in southwestern Ontario have natural flow conditions (i.e., their flow is not controlled by dams) and have a sufficient period of record to permit analysis (Table 1). These watersheds differ greatly in size (from 180 to almost 4000 km^2, in hydrological character (mean annual discharges vary between 90 thousand and 1.7 million dam^3 and in the extent and timing of drainage activity.

For each basin, several hydrological characteristics were analyzed over the period of record from annual summaries:

1. maximum instantaneous discharge
2. maximum daily discharge
3. maximum instantaneous discharge in spring
4. maximum daily discharge in spring
5. date of maximum daily discharge
6. date of minimum daily discharge
7. average daily flow

Each characteristic was analyzed over time and any change for those watersheds experiencing increases in drainage was compared to the extent of drainage and contrasted with those basins with little drainage and with those where drainage has been and continues to be extensive.

RESULTS

Significant differences exist in the amount of extant wetlands in the different watersheds. Evidence from soils data suggests that wetlands were more extensive at one time. Within the study area there is apparently a strong relationship between wetland losses and the extent of drainage. In the absence of verifiable data detailing wetland loss it is convenient to utilize available statistics of expenditures on drainage to represent the loss of wetland areas. These differ from watershed to watershed. However, the impact of these differences, if any, is not established

TABLE 1

GENERAL CHARACTERISTICS OF STUDY BASINS' HYDROLOGICAL RECORD

	Start of Record	Drainage Area (km^2)	Mean Annual Discharge (dam^3)	Discharge (m^3/sec) Extreme Maximum	Discharge (m^3/sec) Recorded Minimum
Ausable River near Springbank	1946	865	306,000	351	0.028
Conestogo River at Drayton	1950	324	114,000	157	0
Ganaraska River near Dale	1950	262	106,000	96.8	0.481
Nith River near Canning	1947	1030	342,000	328	0.453
Nottawasaga River near Baxter	1949	1180	294,000	267	0.878
Saugeen River near Port Elgin	1915	3960	1,770,000	1030	5.72
Sydenham River near Alvinston	1949	730	232,000	207	0.079
Sydenham River near Owen Sound	1945	181	89,900	67.7	0.028

TABLE 2

GENERAL CHANGES IN FLOW CHARACTERISTICS ATTRIBUTED TO DRAINAGE
A REVIEW OF THE LITERATURE

	Annual Runoff	Peak Flow	Spring Flood	Summer Flow	
Baden and Eggelsman (1968)			-		comparative study of a drained and undrained area
Brun *et al*. (1981)	+	+	+		analysis of changes over time at 9 stations
Bulavko and Drozd (1972)	+		+ or -	+	analysis of changes in 6 basins before and after drainage
Burke (1968)		-			comparative study of a drained and an undrained area
Heikuranen (1976)			-	+	comparative study of a drained and an undrained area
Kloet (1971)		+			comparative study of a drained and an undrained area
Mustonen and Seuna (1972)	+		+	+	comparative study of 2 basins before and after drainage
Rannie (1980)		+			analysis of changes over time for 2 basins
Istok and Kling (1981)	-	-			comparison of pre-drainage and post-drainage conditions
Malcolm (1979)		+			inference from hydrological data from a number of sites
Moklyak *et al*. (1972)	-	-	-		comparison of drained and control watersheds
Klueva (1972)	+	-	-	+	comparative study of 16 basins before and after drainage, with 30 control basins

+ indicates that the discharge increased due to drainage
- indicates that the discharge decreased due to drainage

since the results of the analysis were essentially inconclusive. Neither strong evidence nor clear suggestions of any change in streamflow attributable to drainage activity was found. For example, if one considers two watersheds, the Sydenham River at Alvinston (where drainage has been and continues to be very extensive) and the Saugeen River (where significant drainage commenced during the period under study), in neither case can changes in maximum daily flow, average flow, or minimum daily flow be identified and attributed to drainage (Figures 1 and 2).

This is not to say that there were no changes in flow characteristics over the study period. For example, the Sydenham River at Owen Sound (there are two Sydenham Rivers in the study area) has exhibited a regular increase in minimum daily flows since the early 1960s (Figure 3). However, extensive drainage did not begin in this watershed until a decade later, and none of the other watersheds exhibited this pattern. There is no assurance of any consistent influence due to agricultural drainage.

Again, the date of maximum flow in most watersheds has exhibited a greater irregularity in recent years (for example, Figure 4), but this was not associated with substantial changes in drainage activity at the time, and most basins did show this characteristic. It was, apparently, a response to increasing variability of rainfall events.

DISCUSSION

Despite the prevalence of the "sponge" hypothesis, in reality the literature exhibits a wide variety of results from empirical studies of drainage in moist temperate climates where there has been an explicit comparison of drained and undrained conditions (Table 2).

Most notable is that, in contradiction to the "sponge" hypothesis, there is apparent agreement that drainage leads to increases in summer flow. High rates of evapotranspiration in wet areas, especially if they are also heavily vegetated, would produce a reduction in available water at the expense of streamflow. Drainage may, therefore, result in greater flows in summer.

The results regarding peak flow and spring flood are obviously contradictory. At least at times, drainage apparently acts to reduce flow peaks. Figures presented by Burke (1968) and Heikurainen (1976) clearly show this.

These results suggest that the hydrological response to agricultural land drainage is more complex than is often accepted. The negative results described for the watersheds in southern Ontario are perhaps not unexpected:

1. The complex geometry of sub-basins throughout a watershed results in varying effects from drainage projects. One project may result in desynchronization of peak flow, whereas another may add to the peak. Overall, the results will tend to even out the changes in flow. The change by any one drainage project would obviously depend on its position vis-a-vis the rest of the basin.

2. The data used referred to gauges that were all well downstream; if any changes were in fact created upstream, these would be expected to attenuate as they moved downstream.

3. The drainage process is not a single change. Streams are channelized, allowing for more effective removal of water, and fields are underdrained. The hydrologically active layer of soil in many wet areas is shallow. In organic soils, hydraulic conductivity decreases rapidly with depth, sharply limiting vertical penetration of water. This, in addition to the fact that wet areas, by definition, are most often at or near saturation, greatly diminishes the capacity of these areas to substantially reduce stream flow peaks through absorption. Such areas would be poor stream flow regulators except that, topographically, they may allow for the attenuation of flood peaks. On the other hand, once drained, the storage capacity of these areas may be greatly increased through soil storage while the topographic conditions which allow for flood attenuation remain. At a basin-wide scale the net results of channelization and of underdraining are not easily predicated and may cancel one another. In addition, various construction practices and drainage system designs may have very different hydrological effects (Broadhead and Skaggs, 1982; Skaggs and Nassehzadeh-Tabrizi, 1982), which would also tend to suppress any hydrologic changes in an entire watershed.

4. Construction of outlet drains in Ontario is not subject to comprehensive quality control; hence, the drains may be quick to fill in and the transmission of water may be rapidly reduced within a short period after construction.

Agricultural land drainage is an accepted process by which to enhance the productivity of agricultural lands. In Ontario, the process has been widely criticized on administrative, economic, and environmental grounds (Bardecki, 1981; Day et al., 1976; Found et al., 1976). There is no evidence, however, that drainage has led to substantial changes in the hydrological character of the watersheds in Southern Ontario. A review of the literature related to drainage in similar environments suggests that individual projects may produce effects, but, on a broad scale at least, these are not easily predicted and, may indeed counteract one another.

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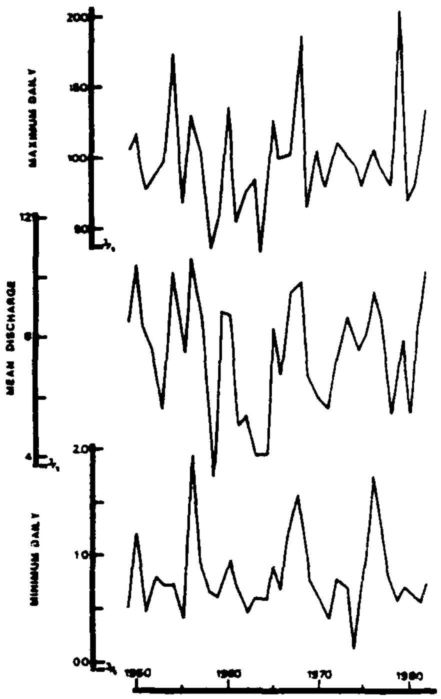


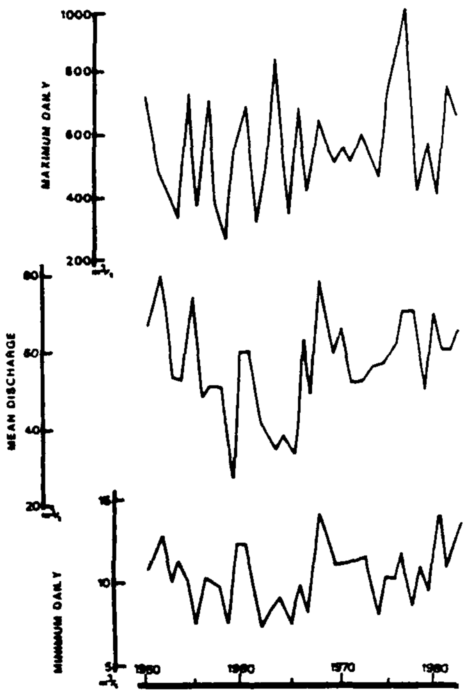
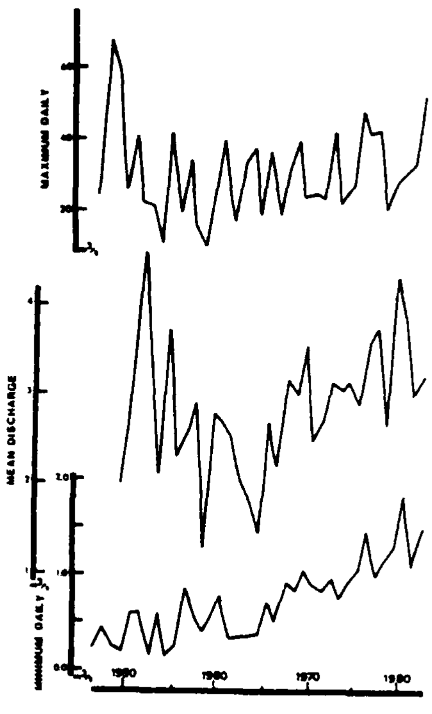


Figure 1: Discharge Record, Sydenham River, near Alvinston.

Figure 2: Discharge Record, Saugeen River near Port Elgin



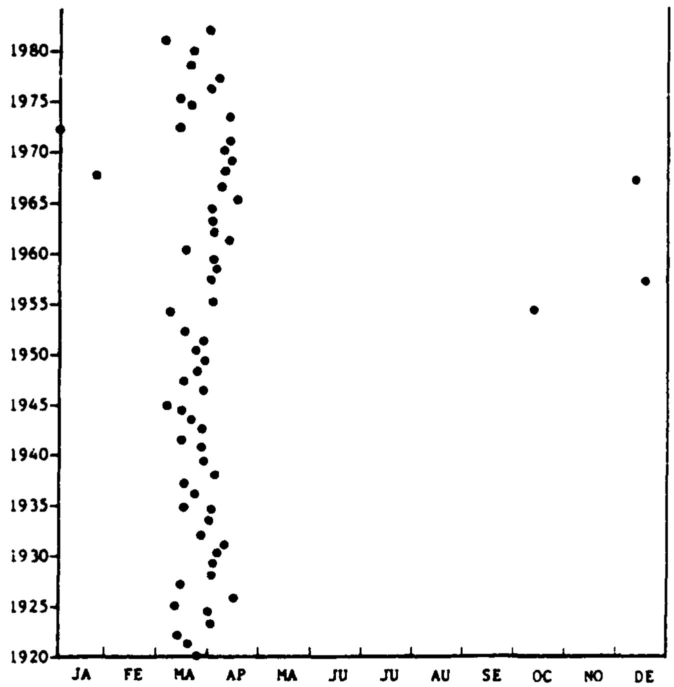


Figure 3: Discharge Record, Sydenham River near Owen Sound

Figure 4: Timing of Maximum Daily Discharge, Saugeen River at Port Elgin

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Effects of Wetland Channelization on Storm Runoff in Lamberts Creek, Ramsey County, Minnesota

R.G. Brown
U.S. Geological Survey

INTRODUCTION

Lamberts Creek, located in Ramsey County near St. Paul, Minnesota, flows into Vadnais Lake from which St. Paul obtains its municipal water supply. During the summer, the water commonly has an undesirable taste and odor that has been linked to algal species associated with eutrophication of the lake (Walker, 1985). The eutrophication is the result of sediment and nutrient inputs from several point and nonpoint sources, including nonpoint sources in storm runoff from Lamberts Creek. Therefore, the quantity and quality of runoff from Lamberts Creek required assessment.

Lamberts Creek channel is a drainage ditch that was constructed initially to drain wetlands for vegetable farming and was used subsequently to drain urban areas. Additional urban development is proposed in the watershed, which may cause additional input of sediment and nutrients to the lake from Lamberts Creek. The Creek is the terminal drainage ditch which drains the channelized wetlands.

Wetland channelization can greatly affect storm-runoff quantity and quality in a watershed by decreasing the surface-water storage capacity and lowering the amount of runoff temporarily stored and by decreasing detention capacity and lessening the capability of the wetland to retain suspended solids transported by storm runoff (Boto and Patrick, 1978; Novotny and Chesters, 1981).

PURPOSE AND SCOPE

Storm-runoff quantity and quality in Lamberts Creek are largely affected by wetlands and other characteristics of the five subwatersheds that constitute the drainage basin of the creek. This paper presents the results of a study that evaluated the effects of wetland channelization on storm-runoff quantity and quality during 12 storms in 1986. The paper addresses the differences in storm runoff by evaluating the amount of wetland channelization and differences in other characteristics of the subwatersheds. The differences in wetland channelization and other basin characteristics are analyzed to determine what effect these factors have on storm-runoff quantity or quality in Lamberts Creek. The selected basin characteristics include land use (including wetland area), impervious area, amount of wetland channelization, basin slope, and channel slope. The interpretation and discussion is limited to storm-related differences in runoff. Annual-related differences in runoff, which include storm and nonstorm runoff, are likely to be different.

METHODS

Figure 1 shows the subwatersheds, data-collection sites, major storm-sewer outlets, and land use. The data-collection sites were placed at the downstream and upstream ends of each subwatershed: subwatershed 1 is the basin between data-collections sites 1 and 2; subwatershed 2 is the basin between data-collection sites 2 and 3; subwatershed 3 is the basin between data-collection sites 3 and 4; subwatershed 4 is the basin between data-collection sites 4 and 5; and watershed 5 is the basin upstream of data-collection site 5.

Lamberts Creek originates as a large storm sewer system in subwatershed 5 that discharges into an open ditch at data-collection site 5 (fig. 1). Lamberts Creek is channelized through all the wetlands except a wetland immediately downstream from data-collection site 5 (fig. 1). The storm-sewered urban area southeast of Goose Lake drains directly into the lake, which has an outlet into the wetland immediately downstream from data-collection site 5.

Basin characteristics of each subwatershed are given in table 1. All the data in table 1, except for basin slope, were obtained from Walker (1986). All land-use and impervious-area data are in km² (square kilometers). Urban land-use areas are those used for residential and commercial purposes. Nonurban land-use areas are those presently undeveloped, excluding wetlands or lakes. Vegetated wetlands are the dominant type of waterbody in the study area. Basin slope, in m/km (meters per kilometer), is the average slope based on an average of 25 basin slopes taken at points on an equal-spaced grid overlaid on a 1:24,000-scale topographic map. Channel slope, in m/km, is the average slope of the main channel determined from elevations at the 10- and 85-percentiles of the distance along the channel within each subwatershed.

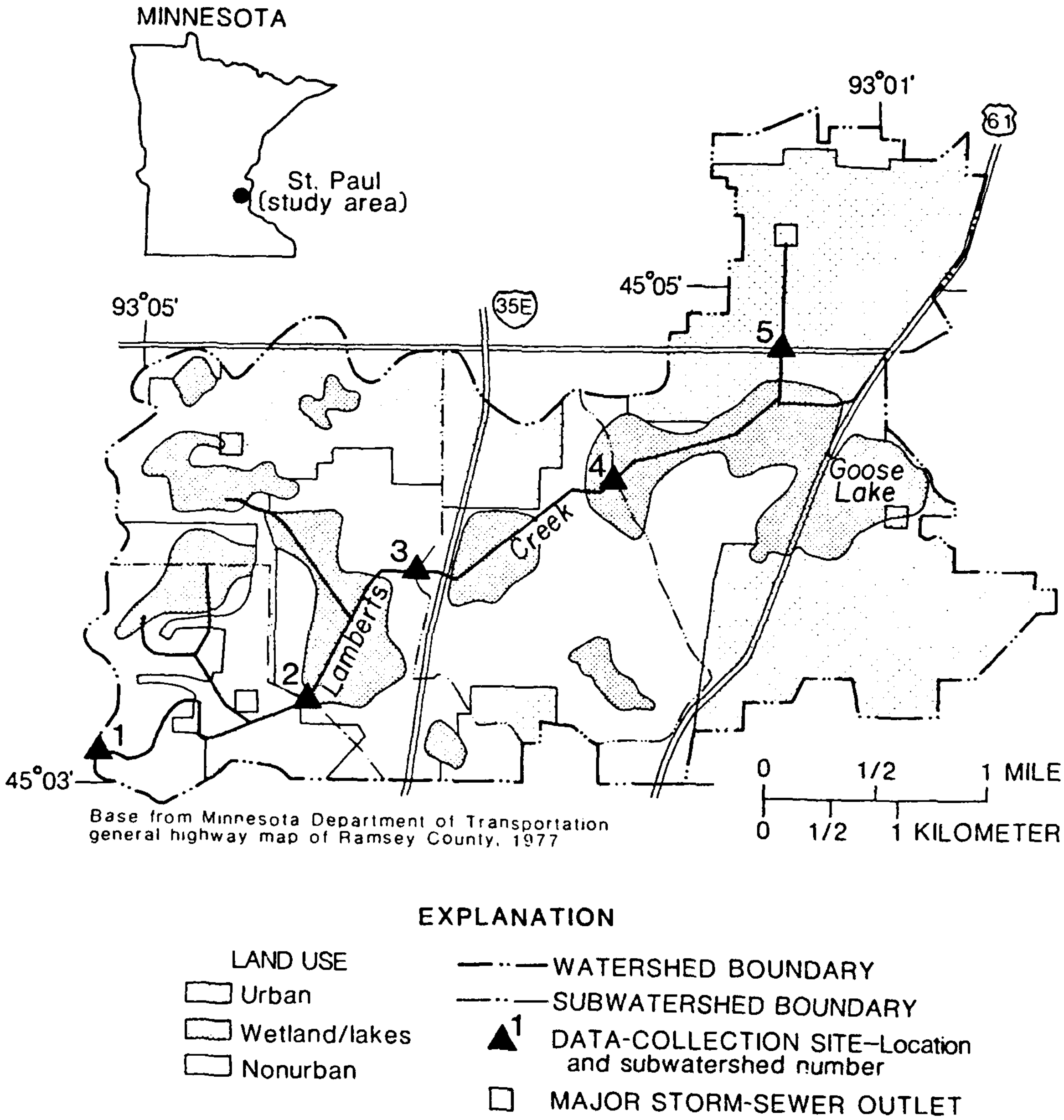

Figure 1.--Location of Lamberts Creek watershed and subwatershed showing land use and data-collection sites.

Table 1. Basin characteristics of the Lamberts Creek subwatersheds (number in parentheses is the percent of basin)

Basin characteristics	1	2	3	4	5
Drainage area (km^2)	2.92	4.80	3.29	5.73	2.77
Urban land use (km^2)	1.32(45)	1.78(37)	1.02(31)	3.01(53)	2.27(82)
Nonurban land use (km^2)	1.09(37)	1.33(28)	1.49(45)	1.44(25)	.45(17)
Wetlands (km^2)	.51(18)	1.69(35)	.70(22)	.76(13)	.03 (1)
Lakes (km^2)	0 (0)	0 (0)	.08 (2)	.52 (9)	0 (0)
Wetlands unchannelized (km^2)	.05 (2)	.19 (4)	.11 (4)	.72(12)	0.00 (0)
Wetlands channelized (km^2)	.46(16)	1.50(31)	.59(18)	.04 (1)	.03 (1)
[percent channelized]	[90]	[89]	[84]	[6]	[100]
Impervious area (km^2)	.46(16)	.69(14)	.33(10)	1.34(23)	.84(30)
Basin slope (m/km)	3.7	10.5	5.5	2.6	.8
Channel slope (m/km)	1.2	.64	2.9	.11	.12

DATA COLLECTION

Storm-runoff quantity and quality data at each of the five data-collection sites were obtained during 12 storms in 1986; one in April, two in May, one in June, one in July, two in August, four in September, and one in October. Storm-runoff quantity was derived from a continuous record of discharge during each storm, and storm-runoff quality was determined by chemical analysis of water samples. Streamflow was calculated from stream-stage data collected every 15 minutes and use of a relation developed between stage and measured discharge (Kennedy, 1984).

Discrete samples for water-quality analysis were collected throughout each storm hydrograph at the date-collection sites by automatic samplers. Storm-runoff quality at each site was determined from a flow-weighted composite sample of the discrete samples collected during the storm hydrograph. Flow-weighted composite samples represent the flow-weighted-mean concentration of the discrete samples, which is calculated from an equation based on the theory of "mid-interval determination of suspended-sediment discharge" (Portersfield, 1972):

$$A_v = (F_s/F_t)S_v \qquad (1)$$

where

$$F_s = q_i t_i \qquad (2)$$

$$F_t = q_i t_i + q_{i+1} t_{i+1} + \dots q_n t_n \qquad (3)$$

A_v = volume of discrete sample to be added to composite sample,

F_s = flow during the discrete sample collection,

F_t = flow during the composite sample collection,

S_v = volume of composite sample required by laboratory,

q_i = discharge (volume per time increment) during collection of discrete sample 1,

t_i = time interval of discrete sample collection which is equal to one-half the time since the previous discrete sample plus one-half the time to the next discrete sample, and

n = the number of discrete samples to be used in the composite.

Composite samples were analyzed for concentration of total suspended solids, total phosphorus, and total nitrogen (ammonia, organic, nitrate, and nitrite) according to methods described by Fishman and Friedman (1985). Storm-runoff loads of each constituent were calculated by multiplying the volume of streamflow associated with the sample (F_t) by the constituent concentration (Nelson and Brown, 1983).

Storm-runoff (streamflow volume per unit area of drainage basin) and storm-runoff yields (storm-runoff load per unit area of drainage basin) from each subwatershed were calculated by subtracting the streamflow volume and load determined at the site upstream from the subwatershed from those determined at the site downstream from the subwatershed and dividing the difference by the intervening subwatershed drainage area. Storm runoff and storm-load yields for each subwatershed are the data used for

evaluating the differences in runoff quantity and quality between subwatersheds as affected by wetland channelization.

Precipitation data were collected every 15 minutes using raingages installed at each data-collection site (fig. 1). The two-axis method (Bethlahmy, 1976) was used to extrapolate point values of precipitation to the average precipitation in the Lamberts Creek watershed.

RESULTS AND DISCUSSION

Effects of Wetland Channelization on Streamflow

The hyetograph, 15-minute total precipitation, and hydrograph, 15-minute discharge, during the first 48 hours of the June storm are shown in figure 2. The figure illustrates the downstream differences in the shape of the Lamberts Creek hydrograph between data-collection sites. The shape of the hydrograph at data-collection site 5 appears to be affected by storm runoff from the impervious area within the highly urbanized subwatershed 5. The shape of the hydrograph includes a large, sharp peak in discharge occurring over a short time, which is typical of urban-runoff hydrographs (Novotny and Chesters, 1981).

The long and flat shape of the hydrograph at data-collection site 4 (compared to data-collection site 5) is likely the result of surface-water storage in the wetland and lake areas within subwatershed 4. The storm-sewer discharge from subwatershed 5 enters the unchannelized wetland in subwatershed 4, disperses throughout it, and is temporarily stored. Storm runoff from the urban area in the southeast section of subwatershed 4 enters Goose Lake, disperses throughout, and is also temporarily stored.

The shape of the hydrograph at data-collection sites 1, 2 and 3 have greater ascending slopes as compared to data-collection site 4 because of storm runoff that enters the channel within each watershed. The stream discharge at data-collection site 3 has a steeper ascending and descending slope in the shape of the hydrograph compared to that of data-collection sites 1 and 2. The shape of the hydrograph for data-collection site 3 is likely to be the result of the steeply sloped channel through the channelized wetlands in subwatershed 3 (table 1). The discharge at data-collection site 3 nearly equals the discharge at data-collection site 4 within 24 hours of the storm, indicating the rapid movement of runoff through the channel.

The increase in stream discharge between data-collection sites 3 and 2 is substantially larger than that between data-collection sites 2 and 1. The larger increase in stream discharge appears to be primarily attributable to storm runoff from the steeply sloped basin and larger drainage basin. The gradual slopes in the hydrographs at data-collection sites 1 and 2 compared to data-collection site 3 is most likely the result of the more gentle channel slopes in subwatersheds 1 and 2.

Effects of Wetland Channelization on Storm Runoff

The total storm runoff during the 12 storms for each subwatershed is shown in figure 3. The total storm runoff during the 12 storms represents 87 to 94 percent of the streamflow that occurred from April through October 1986, depending on the subwatershed. The total storm runoff from subwatershed 5 is greater than that from the other subwatersheds. The larger storm runoff from subwatershed 5 is likely the result of a small wetland area (1 percent of the drainage area; table 1), which minimizes surface-water storage of storm runoff, and a large impervious area (30 percent of the drainage area; table 1), which maximizes the amount of surface runoff.

Storm runoff from subwatershed 4 is the smallest of the five subwatersheds (fig. 3). The large unchannelized wetland area and lake area in subwatershed 4 (12 and 9 percent of the drainage area, respectively; table 1) both provide surface-water storage of storm runoff, which lessens the amount of storm runoff leaving the subwatershed through infiltration and evapotranspiration of runoff in the storage areas.

In comparison, the storm runoff from subwatersheds 1, 2, and 3 are substantially larger than that of subwatershed 4 (fig. 3) and this is probably because the wetlands are 84 to 90 percent channelized (table 1), minimizing the amount of available surface-water storage. Although the impervious area is large in subwatershed 4 compared to subwatersheds 1, 2, and 3 (table 1) the amount of runoff is lower in subwatershed 4 and appears to be the result of the surface-water storage capacity.

Storm runoff from subwatershed 2 is greater than that from subwatersheds 1 and 3 (fig. 3). The greatest difference in basin characteristics between subwatershed 2 and the other two subwatersheds is basin slope (table 1); that of subwatershed 2 is more than twice that of the other two subwatersheds. The rate at which surface runoff moves within the basin is directly proportional to basin slope, all else being equal (Whipkey and Kirkby, 1978). The steeply sloped basin in subwatershed 2 most likely allows for substantially greater storm runoff during the period represented by the hydrograph than that observed in the other two subwatersheds.

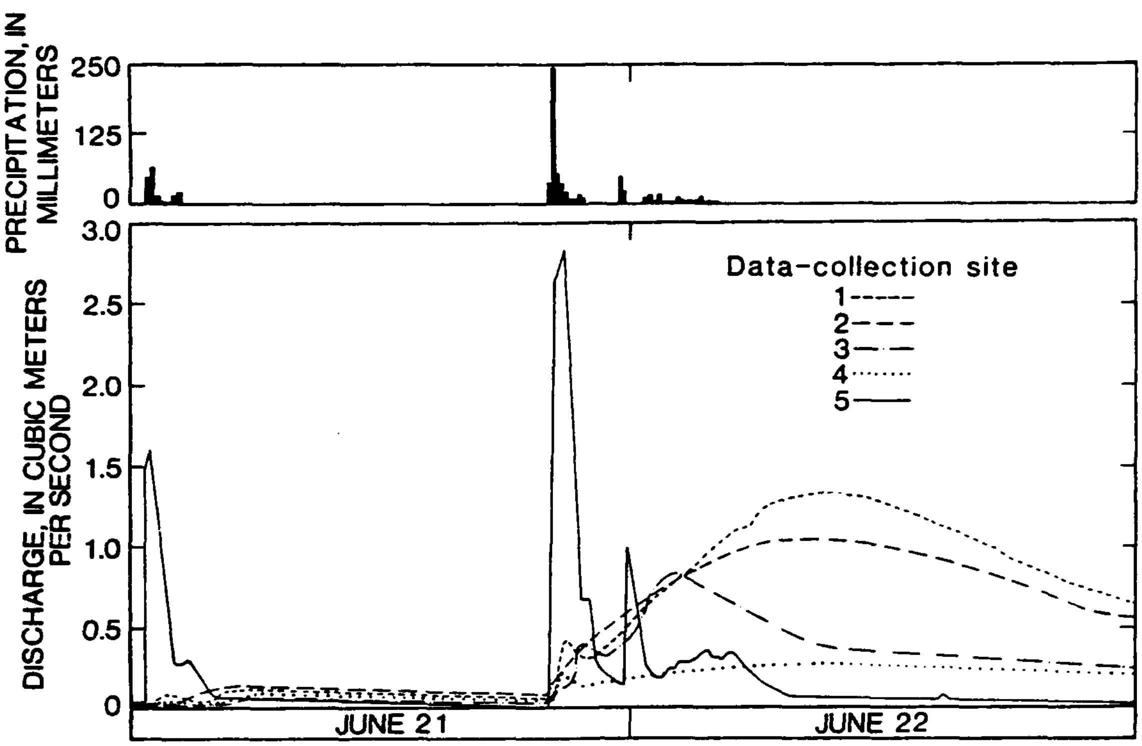

Figure 2. Total 15-minute precipitation and 15-minute discharge of the five data-collection sites during the June 21 storm.

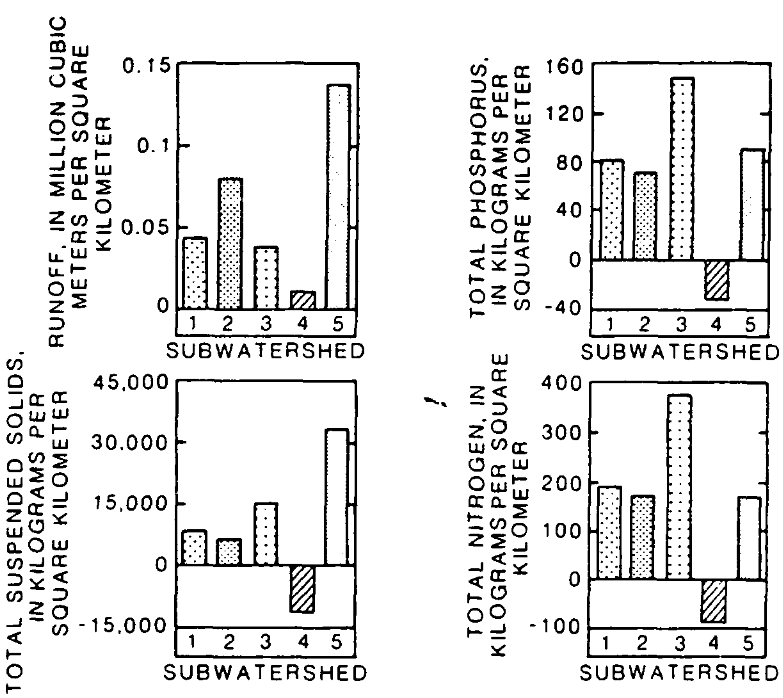

Figure 3. Storm runoff and storm-runoff yields of total suspended solids, total phosphorus, and total nitrogen for each subwatershed during the 12 storms of 1986.

Effects of Wetland Channelization on Storm-Load Yields

Storm-runoff yields of total suspended solids were largest in subwatershed 5 compared to other subwatersheds (fig. 3) and were likely the result of surface runoff from the large impervious area in the subwatershed (table 1). The large amount of surface runoff results in soil erosion, washoff, and sediment transport, generally associated with runoff from urban basins (Novotny and Chesters, 1981). The storm-runoff yields of total phosphorus and total nitrogen were small relative to the other subwatersheds because (1) the washoff from urban basins contains inorganic sediments and, therefore, is low in concentrations of phosphorus and nitrogen and (2) channel erosion in subwatersheds 1, 2, and 3 of highly organic material resulted in large storm-runoff yields of phosphorus and nitrogen (as discussed below).

Storm-load yields of all three constituents leaving subwatershed 4 (at data-collection site 5) for the 12 storms were less than the storm-load yields entering the subwatershed (at data-collection site 4), which appears to be the result of the fact that the wetlands are predominantly unchannelized. The storm-load yields leaving the subwatershed are lower because (1) loads from subwatershed 1 are likely retained in the unchannelized wetland areas and (2) loads from the urbanized area within the subwatershed are likely retained in Goose Lake. Retention of suspended material in the wetland (and lake) is directly related to a decrease in water velocity as the water enters the wetland (and lake). As flow velocity decreases, sedimentation increases (Boto and Patrick, 1978). Vegetation in a wetland tends to decrease water velocity beyond that of pooling alone (such as in the lake) and promotes fallout of suspended material (Fetter et al., 1978). Nitrogen and phosphorus from the suspended material are deposited within the wetland (and lake) and removed from the water column (van der Valk et al., 1978).

The wetlands in subwatersheds 1, 2, and 3 are not effective in retaining suspended material. This appears to be a result of wetland channelization. Storm-runoff yields of suspended solids, total phosphorus, and total nitrogen from subwatershed 3 are larger than yields from subwatersheds 1 or 2 (fig. 3). The large yields are likely to be the result of channel erosion within the channelized wetland areas in the subwatershed. The channel is steeply sloped, nearly twice that of the other two subwatersheds (table 1), and the organic soils in the channel have high concentrations of suspended solids, including phosphorus and nitrogen (Bay, 1969). Although organic soils are present in the channel reaches of subwatersheds 1, 2, and 3, the steeply sloped channel in subwatershed 3 likely results in substantially greater channel erosion.

CONCLUSIONS

Wetlands comprise 1 to 35 percent of the drainage area in the Lamberts Creek watershed, depending on the subwatershed. However, 6 to 100 percent of the wetlands have been channelized for drainage purposes, depending on the subwatershed. The channelization of the wetlands in the Lamberts Creek watershed appears to have affected both the quantity and quality of storm runoff leaving the basin. Storm-runoff quantity is largest in subwatersheds that have (1) predominantly channelized wetlands, which minimize surface-water storage capacity and (2) steep-basin slopes or large impervious areas, which maximize surface runoff and minimize infiltration capacity. The channelization of wetlands, therefore, decreases the amount of surface-water storage available in the wetlands and increases the amount of storm runoff from the subwatersheds. Similarly, storm-runoff loading of total suspended solids, total phosphorus, and total nitrogen is largest in subwatersheds that have (1) predominantly channelized wetlands, which minimize load-retention capacity through sedimentation, (2) steep-channel slopes, which maximize channel erosion and sediment transport through the wetlands, and (3) large impervious areas, which maximize washoff. Subsequently, channelization of the wetlands has likely decreased the amount of load retention in the wetlands and increased the amount of storm-runoff loading from the subwatersheds. In general, the channelized wetlands are not effective, compared to unchannelized wetlands, in storing storm runoff or in retaining loads. Furthermore, channel erosion through the reach of the channelized wetlands appears to be a major source of loads in the subwatersheds, depending on the channel slope and the amount of impervious area.

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Glaciated Prairie Wetlands: Soils, Hydrology, and Land-Use Implications

Daniel E. Hubbard
Department of Wildlife and Fisheries Sciences
South Dakota State University

Jimmie L. Richardson
Department of Soil Science
North Dakota State University

Douglas D. Malo
Department of Plant Science
South Dakota State University

INTRODUCTION

The Prairie Pothole Region (PPR) is a term that has been generally used to refer to the glaciated prairie region of northcentral North America. The PPR extends from central Iowa, through western Minnesota, across both Dakotas (east and north of the Missouri River), across the northern portion of Montana, and well into the provinces of Manitoba, Saskatchewan, and Alberta. Although several kinds of wetlands occur in the PPR, the majority of them are Palustrine wetlands (classified according to Cowardin et al, 1979) situated in closed-basin depressions (Stewart and Kantrud, 1973; Ruwaldt et al, 1979). The wetlands of the PPR are collectively the most important breeding habitat for dabbling ducks (Anas spp.) on the continent (Smith et al, 1964; Bellrose, 1979). Much of the PPR is intensively farmed and represents the western corn belt and the spring wheat region of North America.

Agricultural practices in the Region include the artificial drainage of prairie pothole wetlands both to increase tillable acreage and to eliminate the nuisance of having to plow around the "wet spots" (Leitch and Danielson, 1979). It has been estimated that Iowa has lost 99% (933,453 ha), North Dakota 60% (1,214,100 ha), and South Dakota 35% (283,290 ha) of their original wetland area (Tiner, 1984). The impact of prairie wetland drainage on waterfowl has been discussed in the waterfowl literature for decades. However, despite a considerable amount of literature on the hydrology of glaciated prairie wetlands (e.g., Hubbard, 1981), these wetlands have received little attention in national reviews of the hydrological functions of wetlands (e.g., Carter et al, 1979; Sather and Smith, 1984). The objectives of this paper are to: (1) review the hydrology of prairie pothole wetlands, (2) review the work conducted on the soils of these wetlands, and (3) discuss the hydrologic and land-use implications of the artificial drainage of prairie potholes.

WETLAND CLASSIFICATION

Most prairie pothole wetlands can be classified by Stewart and Kantrud (1971) into 3 types: Class II (temporary ponds), Class III (seasonal ponds), and Class IV (semipermanent ponds). However, Class I (ephemeral ponds), Class V (permanent ponds), Class VI (intermittent alkali ponds), and Class VII (fen ponds) also occur. Zones of characteristic vegetation occur within each of these Classes and are indicative of water regimes. The typical pothole wetland has a zone occupying the central area of the depression (usually the deepest part) that defines the wetland's Class. The zones that occupy the central portion of each Class are: Class I, wetland-low-prairie zone; Class II, wet-meadow zone; Class III, shallow-marsh zone; Class IV, deep-marsh zone; Class V, permanent-open-water zone; Class VI, intermittent-alkali zone; and Class VII, fen zone. The central zone of each Class is typically surrounded (often incompletely) by concentric bands of zones indicative of lesser water regimes. All of these Classes of pothole wetlands are classified in the Palustrine System of Cowardin et al (1979). The wetland zones of Stewart and Kantrud (1971) can be equated to the water regimes of the Cowardin et al (1979) system (Cowardin et al, 1979; Cowardin, 1982). Of these, the most important are: wet-meadow zone = temporarily flooded; shallow-marsh zone = seasonally flooded; and deep-marsh zone = semipermanently flooded. Each wetland Class can be further subdivided into subclasses corresponding to average salinities as indicated

by plant species composition. Space limits further discussion of the Stewart and Kantrud (1971) classification system and the reader is referred to that publication as it will be used throughout this paper.

WATER BUDGET

Pothole wetlands receive their water from either direct precipitation, meltwater from drifted snow, runoff from surrounding uplands, or by groundwater discharge (Shjeflo, 1968; Sloan, 1972; Eisenlohr and others, 1972). Water leaves a prairie pothole by either direct evapotranspiration (ET) from the pond, marginal ET from the pond edge, groundwater recharge, or by overflow out of the depression during high water levels (Meyboom, 1966a; Shjeflo, 1968; Sloan, 1972; Millar, 1971; Eisenlohr and others, 1972). Throughout the PPR average annual precipitation is always less than the average annual evaporation (Omodt et al, 1968; Millar, 1969; Spuhler et al, 1971). Therefore, the amount of water entering a pothole from runoff, meltwater from drifted snow, or groundwater discharge in excess of direct precipitation governs a pond's permanency.

Runoff into potholes occurs during the spring thaw when melting snow or precipitation flows over frozen soil or during the frost-free season when precipitation rates exceed the infiltration capacity of the soil. The glacial till-derived soils of the region are high in smectite clays which expand greatly when wet and are the primary cause of the low permeabilities of the soils. The amount of runoff that a pothole will receive can vary greatly between years, and the relative contribution of snow-melt runoff and growing-season runoff can also vary greatly (Shjeflo, 1968; Millar, 1969). This is also true between localities in the same year. Wide variations in precipitation and temperature are normal in the glaciated prairie region (Omodt et al, 1968; Spuhler et al, 1971).

Other than seepage to groundwater and basin overflow, water leaves a pothole wetland only by ET (Shjeflo, 1968; Allred et al, 1971; Millar, 1971). ET can be separated into that which occurs from the pond <u>per se</u> and that from the pond's edge (marginal plants and soil) (Millar, 1971). Pond marginal ET is important in all pothole wetlands in terms of soil formation, but is quantitatively most important to the water budgets of small wetlands. Millar (1971) attributed from 60 to 80% of the water loss from ponds less than 0.4 ha to marginal ET, but not more than 30 to 35% of total loss in ponds larger than 0.4 ha. In his study, the rate of water loss varied directly with the length of shoreline per unit of pond surface area, and although only ponds of about 4.0 ha or less were studied, it would only seem logical that as ponds become very large the effect of marginal ET would become a smaller part of total water loss. Even though the effect of emergent hydrophytes on ET rates is variable (Bernatowicz et al, 1976; Idso, 1981), it was found that vegetated potholes in North Dakota annually lost less water via ET than did open water potholes (Eisenlohr, 1966; Shjeflo, 1968). This effect is caused by the sheltering of the water surface by senescent plants at the beginning and end of the growing season.

GROUNDWATER RELATIONS

There are 3 general types of pothole wetlands in regard to groundwater (Lissey, 1971; Sloan, 1972; Winter and Carr, 1980): groundwater recharge wetlands, groundwater discharge wetlands, and flowthrough wetlands that both recharge and discharge groundwater at various locations within the pothole. Depending on fluctuations in the watertable, a pothole may temporarily change from one type to another (Lissey, 1971; Winter and Carr, 1980). The degree to which groundwater discharge takes place in a pothole wetland is roughly related to its salinity, and therefore, its conductivity (Rozkowski, 1969; Sloan, 1972). Those with the freshest water are recharge wetlands and those with very saline water are discharge wetlands. Flowthrough wetlands, however, are intermediate in salinity. The higher salinities in discharge wetlands are a result of evaporative concentration of salts with no mechanism for their removal (i.e., no downward movement of water). Major ions responsible for the salinity differences are Na^{+}, Mg^{2+} and SO_4^{2-} (Rozkowski, 1969; Stewart and Kantrud, 1972; Arndt and Richardson, 1986).

Water conductivity measurements at a point in time are generally not reliable for determination of groundwater regimes. Salinities fluctuate seasonally, tending to be lowest in spring and highest later in the season due to concentration of salts at low water levels. Large runoff events can dilute the pond water at any time. Variations in chemical composition of the soils and drift can affect the background conductivity of the groundwater. These factors can cause local differences in groundwater conductivities such that general levels in pond waters may vary between areas.

Groundwater flow systems in the PPR are of 3 general types: local flow--where groundwater moves between adjacent potholes; intermediateflow--where groundwater may move at deeper depths and discharge into potholes not adjacent to the recharge source but still in the local area; or regional flow--where groundwater moves deep into the till and interacts with wetlands in distant areas (Toth, 1963; Lissey, 1971; Winter and Carr, 1980). The major systems interacting in regional topographic highs (e.g., the "knob and kettle" or "hummocky moraine" areas of dead ice moraine) are typically local and intermediate, while those in regional topographic

lows (e.g., ground moraine) may receive groundwater from regional flow systems that originate in adjacent dead ice moraine (Meyboom, 1966b; VanVoast and Novitzki, 1968). Factors influencing which system a wetland is interacting with depend on the topographic setting, position of the water table, thickness of the drift, anisotropy of the drift, and the configuration of underlying bedrock (Freeze and Witherspoon, 1967; VanVoast and Novitzki, 1968; Winter and Carr, 1980).

The classification of prairie pothole wetlands can be related roughly to groundwater relationships. Based on the information provided by Lissey (1968, 1971), Sloan (1972), Winter and Carr (1980), Richardson and Bigler (1984), Miller et al (1985), Arndt and Richardson (1986), and Hubbard et al (1988), it may be generally stated that groundwater recharge wetlands are typically Class II and III, discharge wetlands are typically Class VI and VII, and the saline subclasses of Classes IV and V. Flowthrough wetlands are typically Class IV and V, but some may also be Class III and VII.

POTHOLE WETLAND SOILS

The movement of groundwater within the landscape of the PPR governs the transfer of readily soluble ions from points of groundwater recharge to points of discharge. The movement of these ions and the permanence of water at the surface affects soil development by controlling the soil profile processes (Miller et al, 1985; Arndt and Richardson, 1986). In Class II and III wetlands, the dominant flow direction is downward (Richardson, 1986; Knuteson et al, 1987a). Water leaches the readily soluble ions from the soil. If the weathering products can be removed, hydrolysis of the primary minerals in these wet environments is at a maximum. Such removal results in formation of clay. Additionally, the wetting and drying cycles result in clay eluviation from the soil surface downward to create an argillic horizon. In the PPR, wetlands with recharge characteristics have the most developed soils. Depths to the bottom of the solum are typically up to 2 m or more in recharge potholes (Miller et al, 1985; Hubbard et al, 1988). The contrast of these wetland soils with unleached soils is one between nearly opposite types, i.e., most developed and least developed soils (Fulton et al, 1986).

Class IV and V wetlands are either flowthrough or groundwater discharge wetlands, and most probably shift between the 2 types depending on watertable fluctuations. After periods of drought when many Class IV wetlands are dry, they may even function temporarily as recharge wetlands until watertable levels rebound. These wetlands typically have a combination of soil characteristics indicative of both groundwater recharge and discharge. The net effect is that water does not leach the soluble ions away and the soils lack the influence of consistentlydownward water movement through the soil profile. The permanence of the water allows for stagnation and chemical reduction. These soils have a build-up of readily soluble ions, an increase in soil organic matter, lack clay translocation, and are subject to reduction (Richardson and Bigler, 1984; Miller et al, 1985). They have a thick A-horizon and usually lack a B-horizon (Bigler and Richardson, 1984; Miller et al, 1985).

Wetlands that are predominantly groundwater discharge sites are Classes VI and VII and the most saline subclasses of Classes IV and V. The soils are usually simply sediments with or without a thin A-horizon (Richardson and Lura, 1986; Richardson, 1986). Last (1984) treats saline wetlands of this type extensively. The fens (Class VII) are special cases of discharge sites that accumulate peat and have not been studied.

At the edge of most pothole wetlands studied, water moves upward from the watertable to the surface due to pond margin ET effects. This area along the edge of wetlands has long been known to be important in accumulation of evaporite minerals (Whittig and Janitzky, 1963; Redmond and McClelland, 1959). Large amounts of calcium carbonate (Knuteson et al, 1987a and b) and gypsum (Steinwand and Richardson, 1987) occur here. The tendency is for more water to move upward via ET than is added from above. The net result is the accumulation of evaporites. The first mineral to form is calcite ($CaCO_3$). If calcium is more abundant, gypsum forms next. However, if carbonate is more abundant, several minerals form, such as protodolomite (Hardie and Eugster, 1970; Rostad, 1975), burkeite (Jones, 1965; Keller et al, 1986), and various other sodium carbonates (Whittig and Janitzky, 1963). Two soils result: Calciaquolls (Knuteson et al, 1987b) and Natraquolls. The latter are not as productive as the former due to the presence of sodium.

LAND-USE AND HYDROLOGIC IMPLICATIONS OF ARTIFICIAL DRAINAGE

Soil Salination

Groundwater recharge sites (Class II and III wetlands) have some of the most highly developed soils of the prairies; sola are very deep, are leached free of soluble salts and carbonates, and have argillic horizons. If these soils are artificially drained, they can produce excellent yields of cultivated crops. However, the artificial drainage and cultivation of groundwater flowthrough or discharge wetlands (Classes IV to VII) can cause soil salination (Richardson, 1986). The soils in discharge wetlands are saline to begin with, but those of flowthrough wetlands are

only moderately saline at the surface. After artificial drainage and cultivation of a flowthrough wetland, groundwater is no longer transpired from depth in the soil by hydrophytes and is allowed to evaporate directly from the soil surface. Therefore, salts that accumulated at depth are now free to accumulate at the soil surface. This same process can occur on the margins of flowthrough or discharge wetlands that are not artificially drained, but have their marginal Calciaquoll (or Natraquoll) cultivated (Richardson, 1986).

Groundwater Implications

Effective recharge to the groundwater in these glaciated landscapes has been theorized to be depression focused (i.e., occurs only where water is ponded, as in wetlands) (Freeze, 1969; Freeze and Banner, 1970; Lissey, 1971). If this is true, then widespread artificial drainage of wetlands, regardless of type of groundwater interactions, should eventually lead to general watertable decline. Consequences of lowered watertables could be lowered heads in domestic wells located in buried outwash in the till and possibly a circumvention of soil moisture recharge from groundwater. However, no studies have been conducted to address these issues.

It has been shown that over the course of winter, water can move upward from the watertable to the frozen soil above it in response to thermal gradients in the till soils of the PPR (Schneider, 1961; Willis et al, 1964; Benz et al, 1968; Malo, 1975). Malo (1975) reported soil moisture increases due to this phenomenon of greater than 10% at the 60 cm depth at a summit position 80 m from a Class III wetland in eastern North Dakota. Watertable depths dropped from about 3.7 m to 4.3 m below the soil surface during this soil moisture recharge event. The proportion of the landscape in the PPR that receives soil moisture recharge from the watertable is not known, but we speculate that it is substantial. If watertables decline due to artificial drainage of potholes, and this source of soil moisture is precluded, then negative economic impacts of potentially large proportions may occur to crops in this drought-prone region.

Surficial Drainage Patterns

Conclusive documentation of the effect of artificial drainage on flooding problems in the PPR has not been published (Linder and Hubbard, 1982). However, computer simulation studies (Campbell and Johnson, 1975; Dybvig and Hart, 1977; Moore and Larson, 1979) and empirical studies (Kloet, 1971; Moore and Larson, 1979; Rannie, 1980; Brun et al, 1981) provide compelling evidence that the artificial drainage of wetlands in the PPR has probably had major contributory effects on flooding problems in the region in recent decades. The amount of water that can be collectively stored in potholes over an area is large (Ludden et al, 1983; Hubbard and Linder, 1986). Likewise, the amount of noncontributory area in PPR watersheds can be impressive, certainly in the millions of acres, especially under dry conditions. Stichling and Blackwell (1957) have described the fluctuating drainage area phenomenon in detail and provide an example of a watershed that under dry conditions (depression storage empty) had a net drainage area of 20% of the net drainage area under wet conditions (depression storage full and wetlands overflowing). If a depressional watershed were to be completely "ditched-out," then the net contributing area will be permanently increased to the size of the "net wet drainage area." The relationship between increasing drainage area and increasing watershed discharge has long been recognized by hydrologists (Strahler, 1964). While the magnitude of the largest flood events may not be changed from the natural condition in an artificiallydrained PPR watershed, it would seem logical to predict that the magnitude of smaller flood events may increase as would the frequencies of all flooding events. The data presented in Kloet (1971), Moore and Larson (1979), Rannie (1980), and Brun et al (1981) reveal that these consequences of artificial drainage may indeed have been realized.

CONCLUSIONS

Much is known about the hydrology of prairie pothole wetlands. They are integral parts of both the surficial and subsurface prairie hydrological systems. The hydrology of each wetland determines the character of the soils in depression centers. Soils at depression rims are influenced by ET at the pond margins, creating an environment of evaporite deposition. The significance of groundwater recharge to the maintenance of watertable height, especially at higher regional elevations (i.e., in regional recharge areas) needs to be studied and the importance of soil moisture recharge from groundwater needs to be adequately addressed. Innovative approaches to the problem of evaluating the impacts of wetland drainage on flooding problems in the PPR need to be designed and implemented. Soil salination due to the drainage of Class IV wetlands needs to be quantified in terms of long-term losses of both agricultural productivity and overall biological productivity once a salinified basin is restored (i.e., the ditch plugged). The evidence for prairie pothole wetlands being integral and important components of the prairie hydrological regime is so overwhelming, however, that while society waits for definitive studies to document the true value of these wetlands, the resource may be drained and lost, and a significant portion of the agricultural productivity of the region may be lost as well.

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Hydroperiod Prediction for Four New Palustrine Wetlands

Donald L. Raisanen, Steven D. Leimer, G. Nicholas Textor
Envirodyne Engineers, Inc.

INTRODUCTION

Rapid urbanization in DuPage County, Illinois required construction of a major north-south expressway. In 1984 the Illinois State Toll Highway Authority (ISTHA) proposed the 17.5 mile North-South Tollway. The proposed corridor would result in the direct loss of 76.2 acres of wetlands and indirectly impact an additional 268.8 acres of wetlands adjacent to the construction. This necessitated a U.S. Army Corps of Engineers (USACE) Section 404 permit. Conditions of the permit required 120 acres of new wetlands to be constructed as mitigation. Envirodyne Engineers, Inc. (EEI) was retained by ISTHA to design four new Palustrine wetland areas for this purpose. The wetland sites were to be located along the East Branch DuPage River which would serve as the primary input source of water.

Four locally important wetland habitats were chosen to be incorporated into the design of the wetland sites. These habitats were open water, erect emergent, wet prairie/sedge meadow, and wet mesic prairie. The habitats required flooding at least once during the growing season. In addition, water circulation was desirable in order to provide flushing.

Because the relationship of water levels over time was essential to wetland habitats, hydroperiod prediction was a critical design factor. While much data was available for high flood flows, the required water elevation data at normal and lower flood flows was lacking. This paper presents the methodology used to predict the hydroperiod for these lower flows. It also describes why design water levels were chosen at each site.

METHODOLOGY

General

The hydroperiod prediction utilized three methods of analysis consisting of hydrologic and hydraulic computer modeling, examination and evaluation of existing stage discharge data collected by others, and collection and evaluation of field surveyed data.

Method I-Computer Modeling

As part of the expressway design, a hydrology and hydraulic analysis was performed to determine the effect of the proposed North-South Tollway on flood conditions in the tollway corridor. The basis of the analysis was the development of a hydrology model to predict peak flows from the East Branch DuPage River watershed and a hydraulic model to predict water surface elevations at various points along the East Branch DuPage River (Envirodyne Engineers, Inc., 1986). To perform the hydroperiod prediction, these models were modified to predict flows and water surface elevations at the four wetland sites.

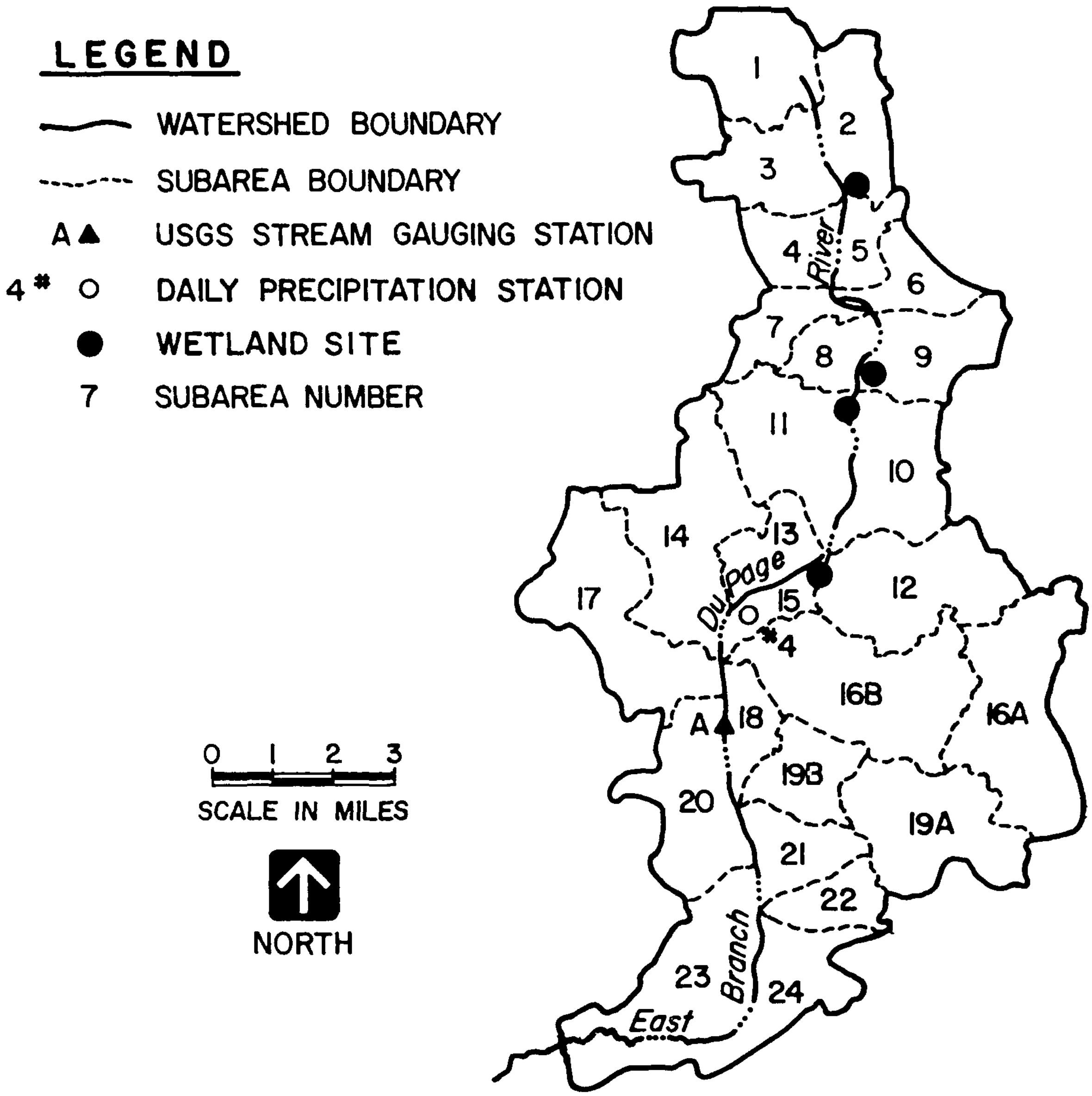
Hydrology Model

Hydrologic modeling was done using the USACE Flood Hydrograph Package (HEC-1), January 1985 version (Hydrologic Engineering Center, 1981). The HEC-1 model was used to predict flows for the 79.4 square mile East Branch DuPage River watershed shown in Exhibit 1.

The model utilized existing data collected by others. Field work was limited to general field reconnaissance surveys to verify existing data. Precipitation data for the model was taken from a gage located within the watershed. The gage had been in continuous operation for a 49 year period of record prior to the hydrology analysis. A model calibration to a historical flood producing event was performed. Flows at the four wetland sites were generated for the 1-, 2- and 5-year recurrence interval flood events. Statistical limitations prevented the calculation of flows below the 1-year event. However, measured flow data was available from a water quality study performed by the U.S. Geological Survey (USGS) in July 1983 (U.S. Geological Survey, 1986). These flows represented an event below the 1-year event.

Hydraulic Model

Hydraulic modeling was done using the USACE Water Surface Profiles (HEC-2) model, January 1981 version (Hydrologic Engineering Center, 1981). The HEC-2 model was used to predict water surface elevations along the East Branch DuPage River assuming steady gradually varied flow. The effects of various obstructions



THE E. BRANCH OF THE DUPAGE RIVER WATERSHED

EXHIBIT I

such as bridges, culverts, weirs, and structures in the flood plain were considered in the computations.

Previously collected river cross section geometry was supplemented by field surveyed cross sections taken adjacent to the four proposed wetland sites. Results of the hydrologic modeling in the form of flows at various points along the East branch DuPage River were input to the hydraulic model. Water surface profiles were generated for the 1-, 2-, and 5-year recurrence interval flood events. In addition, water surface profiles were generated for the July 1983 event and for flow averages of the 1- and 2-year storms and 2- and 5-year storms.

Method II-Stage Discharge Data Analysis

Reliable stage data existed for only one gaging station on the East Branch DuPage River located at Lisle, Illinois, River Mile (RM) 38.53. This gage is operated and maintained by the USGS. The available record included nearly 24 years of mean daily gage heights. Statistical computation of the mean water surface elevation at Lisle was determined by averaging all mean daily gage heights below that of the 1-year flood event for the period of record. A similar computation of mean water surface elevation for the growing season was defined as the period of April to October, inclusive.

As previously mentioned, the HEC-2 model was used to generate water surface profiles along the East Branch DuPage River. This enabled a relationship to be established between flood elevations at the Lisle gage, RM 38.5 and the four wetland sites at RM 42.4, 45.0, 45.4, and 49.6. A family of curves was developed with elevations at Lisle plotted along the abscissa and wetland site elevations plotted along the ordinate. Lines intercepting the mean elevation at Lisle, as determined by the statistical computation were read off the ordinate to determine projected mean elevations at the four proposed wetland sites. Exhibit 2 shows an example of one of these curves.

The stage data was also used to check the frequency of storm occurrence as predicted by computer modeling. This was accomplished by comparing the known stage elevations with the computer generated 1-year event.

Method III-Field Survey

The third method of analysis was a field survey to determine the elevation of the streambank vegetation. It was assumed that during the growing season certain streambank vegetation would be limited by the water surface elevations. Reed canarygrass, Phalaris arundinacea, was selected as the indicator species since it does not grow in standing water. In the field, the elevations of the root collar of the reed canarygrass were measured on the streambanks at all of the sites.

Other survey data included groundwater elevations obtained from soil borings. Approximately 15 borings were taken at each site for construction purposes. Groundwater elevations were measured at the time of the boring and 24 hours after the boring.

Results

Each of the three methods of analysis produced a set of low flow elevations from which design water levels could be chosen. Exhibit 3 shows the relationship of the data from the three methods of analysis at the Lisle gaging station. Exhibit 4 shows the relationship of data at four wetland sites.

The computer modeling produced elevations at the Lisle gaging station which were 2.5 feet, 4.7 feet and 8.1 feet above the stream bottom for the July 1983 event, the 1-year event and the 2-year event respectively. The computer generated profiles at the wetland sites generally paralleled the slope of the river bottom except at the East Branch Site (RM 49.6). At this site, the water surface elevations were nearly level.

The stage data produced a mean elevation which was 2.0 feet above the stream bottom at the Lisle gaging station. It is noted that the mean elevation computed for the growing season was equal to that computed for all seasons. The water surface profile produced by projecting the mean elevation upstream (as determined from the family of curves) paralleled the computer generated profiles at the three wetland sites nearest the gaging station. The mean was below the July, 1983 level at the Hidden Lake site (RM 42.4) and above the July, 1983 level at the Roosevelt Road site (RM 45.0) and the Route 53 site (RM 45.4). The projected mean at the East Branch Site was found to be below the river bottom.

The comparison of the 1-year flood event elevation at the Lisle gaging station to the gage heights revealed the following information.

1. The 1-year flood event elevation was reached or exceeded in all years except one: 1977.

2. The 1-year flood event elevation was reached or exceeded during all but five growing seasons: 1962, 1971, 1974, 1977, and 1984. It is noted that the 1-year event elevation was reached in March of 1962, February and December of 1971, and February 1984. It is also noted that the 1974 records were available only from August 1 to December 31.

3. The water surface elevation exceed the

PROJECTED MEAN GRAPH

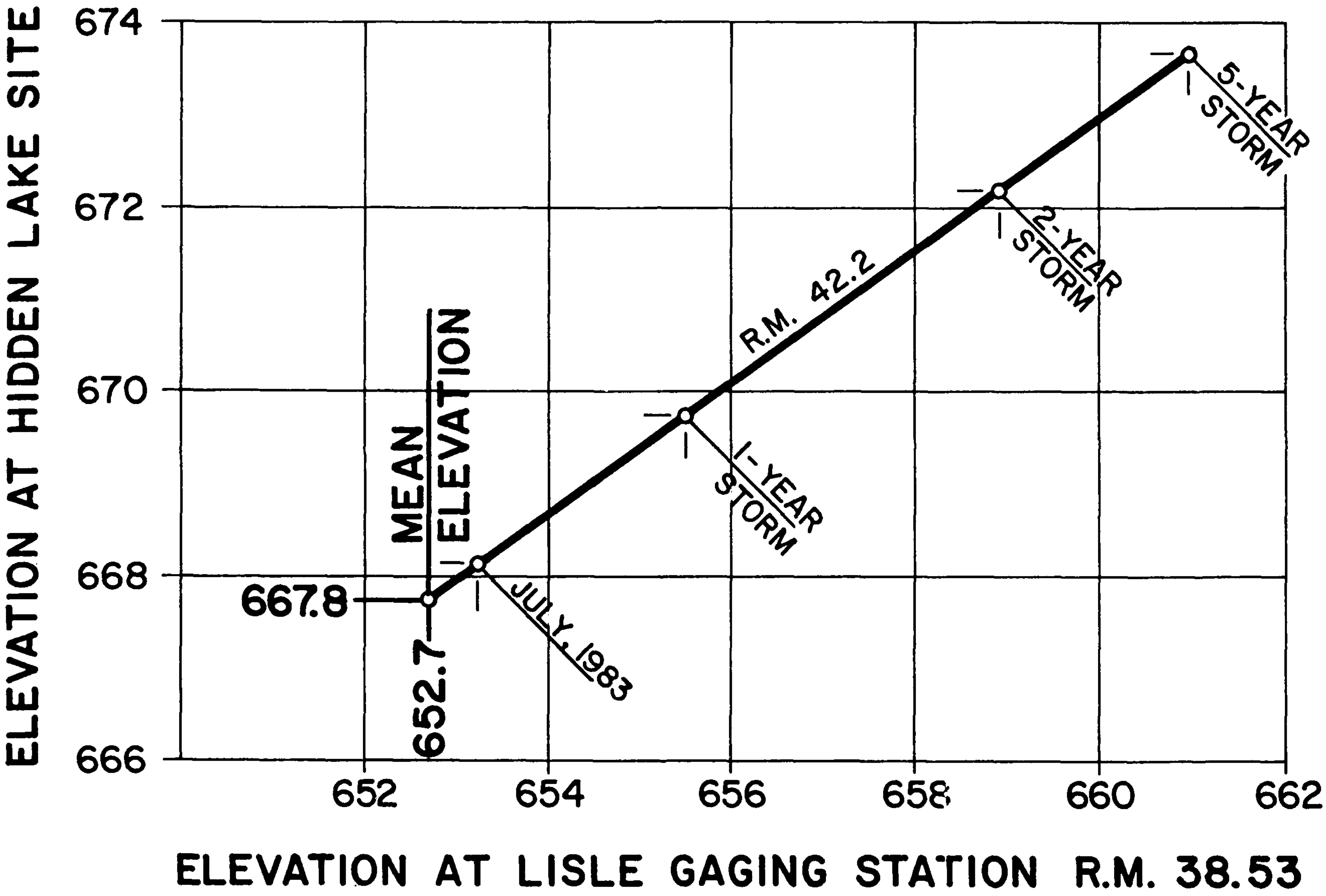
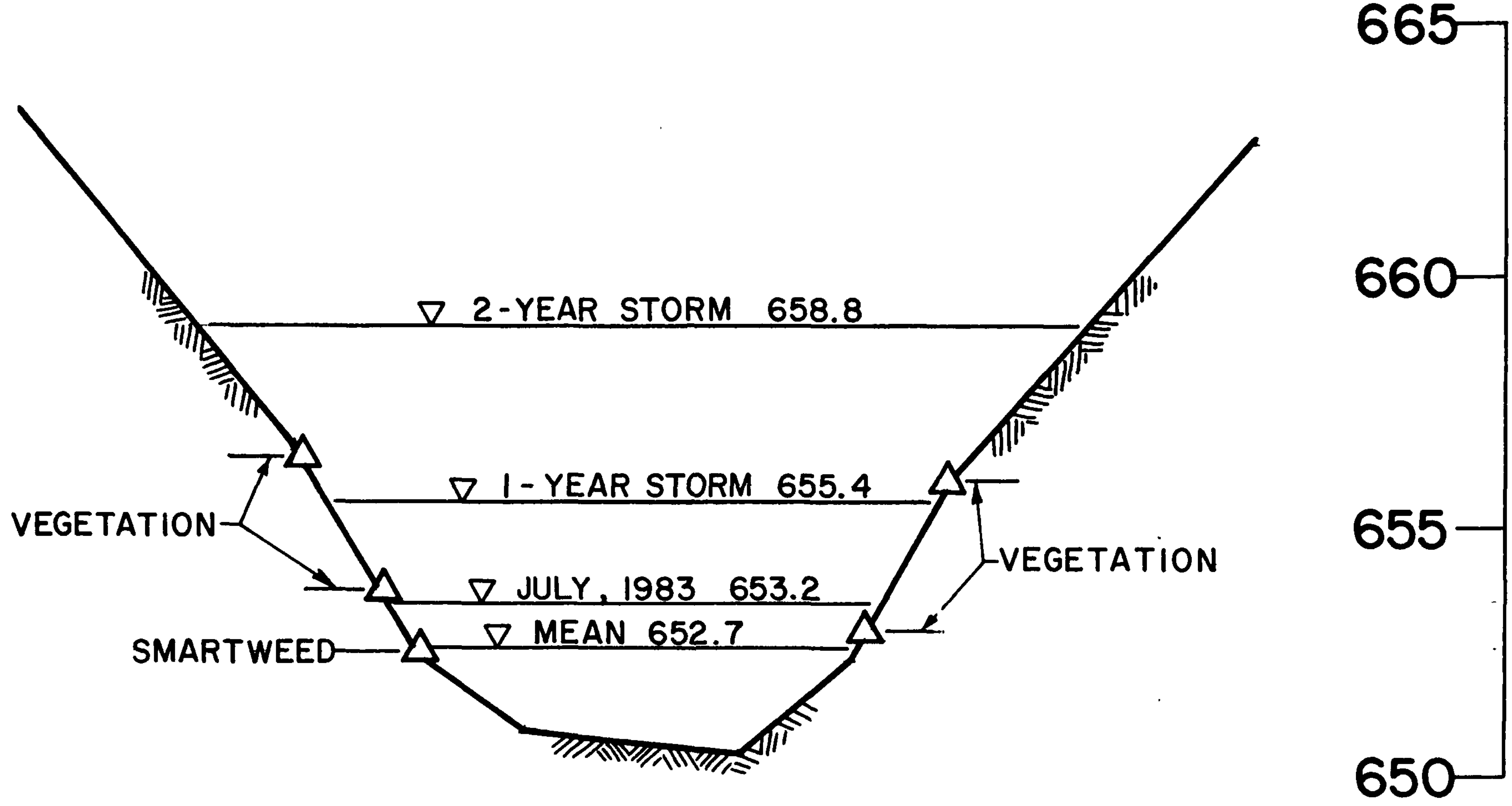
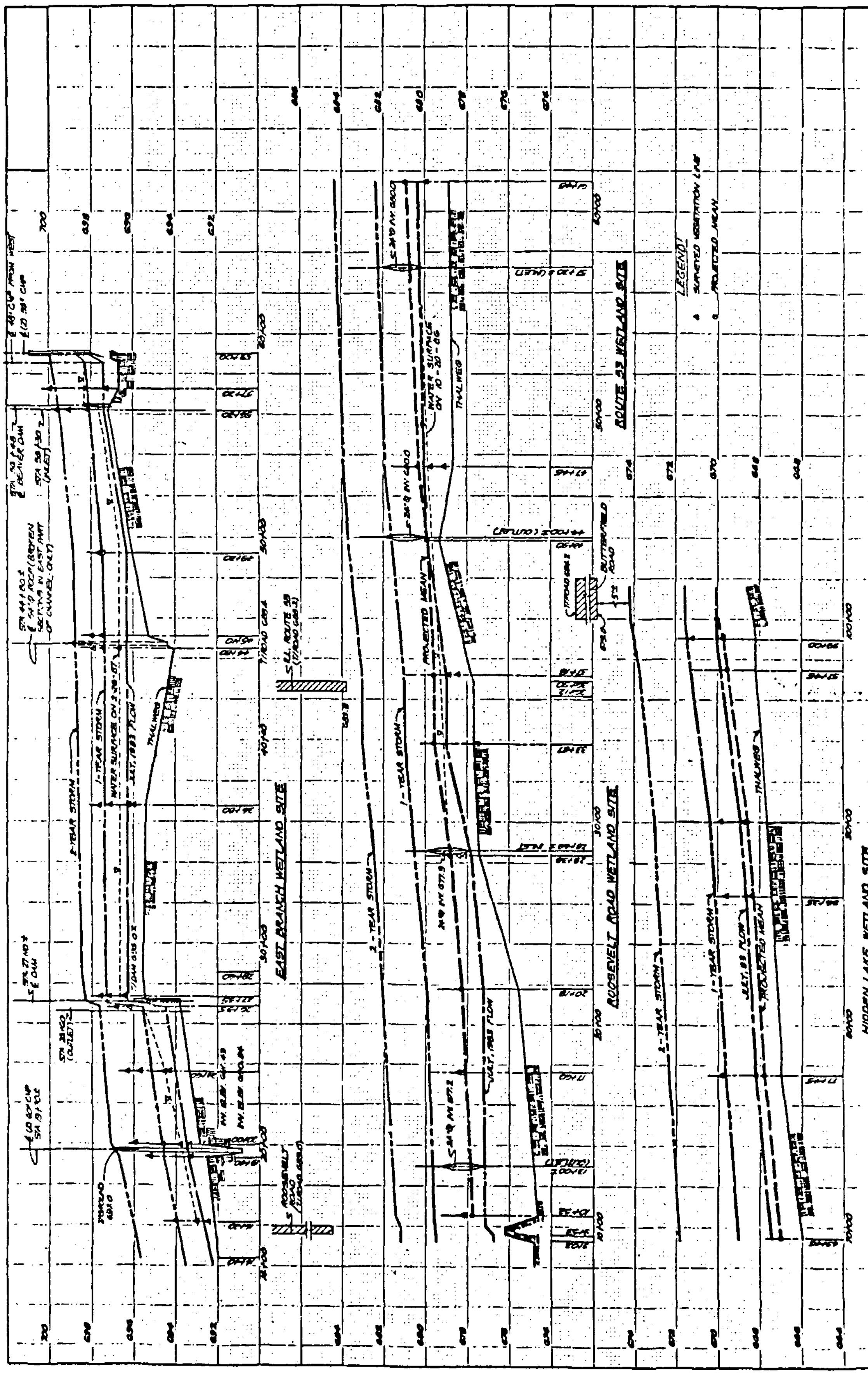


EXHIBIT 2



EAST BRANCH DuPAGE RIVER
CROSS SECTION AT LISLE GAGING STATION
EXHIBIT 3



HYDROPERIOD ANALYSES

EXHIBIT 4

1-year event elevation during the growing seasons in 1960 and 1961. However, it was below the 1-year event elevation for a 17 month period from April 19, 1960 to September 12, 1961.

4. In 1964, the water surface elevation at Lisle was above the 1-year elevation on only one day.

The surveyed elevations of the root collars of the reed canarygrass varied between the projected mean elevation and a point slightly above the 1-year storm elevation at Lisle and the Hidden Lake site. At the Roosevelt Road site, the vegetation was approximately 1 foot below the projected mean. The elevations of the root collar at the East Branch site varied between the 2-year and the July, 1983 events.

The average groundwater elevation measured during the soil borings was considered to be uniformly level across the sites. At the Hidden Lake site (RM 42.4), the groundwater elevation varied from 0.5 feet above to 0.5 feet below the 1-year flood elevation. At the Roosevelt Road site (RM 45), the groundwater elevation varied from 0.2 feet below to 0.5 feet above the projected mean. At the Route 53 site (RM 45.4), the groundwater elevation varied from 0.8 feet below the projected mean to 0.2 feet above the projected mean. At the East Branch site (RM 49.6), the groundwater elevation was approximately equal to the July, 1983 elevation.

Discussion

The following discussion described the physical conditions of each site in relation to the range of elevations produced by the three methods of analysis. It explains how design elevations were chosen for each site.

Results of the comparison of the 1-year flood elevation to gage heights at Lisle indicated that the 1-year flood elevation may not occur frequently enough to meet the design criteria which required circulation and annual inundation within the wetland. It was reasoned that an elevation closer to the mean would provide an acceptable amount of water flushing for supporting the plant communities.

The Hidden Lake site (RM 42.4) required that the wet prairie/sedge meadow habitat be placed no more than 6 inches above the normal river water level. No control structures were to be used and frequent inundation was required. Exhibit 4 shows that the mean elevation was 6 inches below the July 1983 level, similar to the Lisle gaging station. Because the Hidden Lake site is the closest to the Lisle gaging station, the projected mean was determined to be the most accurate at this location. Therefore, the projected mean elevation was chosen as the design water level.

The Roosevelt Road site (RM 45) utilized control structures on one side of the river but none on the other. The design water level for the controlled side was set 6 inches above the projected mean elevation in a similar fashion to the Hidden Lake site. However, the projected mean was higher than the July 1983 flow, which indicated that the wetland might be inundated less frequently than at the Hidden lake site. This was desirable because it created a different wetland which could be studied and compared to the others. The elevation for the controlled side was set to allow the wetland to drain or to protect it from flooding during the 1-year event. This was accomplished by setting the top of the control structure at the 1-year event elevation and would optimize management of the wetland.

Conditions at the Route 53 site (RM 45.4) required that water be transported 200 feet from the river to the wetland. This would be accomplished through the use of swales or ditches. However, the ditches crossed a large diameter gas main which could not be relocated or lowered. The design water wetland level was therefore determined by physical conditions. Fortunately, the inlet elevation could be placed slightly below the projected mean elevation.

The East Branch site (RM 49.6) was the farthest site from the Lisle gaging station and was located near the headwaters of the East Branch DuPage River. Projected mean elevations were found to be below the stream bed and discarded. A small dam existed near the downstream portion of the site causing backwater which was nearly level for the length of the site. Computer modeling indicated that the dam controlled the water surface elevation. Vegetation lines were found to be within the normal range. The design water level was chosen to be equal to the top of dam elevation. This would assure that the river level would be slightly higher than the wetland design level during low flows. Therefore, good water circulation and flushing would be provided.

In summary, three methods of analysis were used to determine low flow elevations. These elevations were compared to each other and to varying physical conditions in order to determine a design water level for the wetlands.

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Forested Buffers for Overbank Flow Velocity Reduction Along the Lower Mississippi River

Charles V. Klimas
U.S. Army Engineer Waterways Experiment Station
Vicksburg, Mississippi

INTRODUCTION

The US Army Corps of Engineers, Lower Mississippi Valley Division, is responsible for construction and maintenance of flood protection levees, bank stabilization revetments, and other structures along the Lower Mississippi River. Forest clearing within the leveed floodplain has resulted in overbank flow velocities sufficient to produce scouring and extensive sediment deposition in some areas adjacent to the river. Monitoring of flow velocities at one cleared site (US Army Engineer District (USAED), St. Louis, 1985) and observations at other sites suggested that reestablishment or maintenance of forested buffer zones along the riverbanks may be important to levee integrity and bankline stability during floods. The field survey reported here was designed to provide information on the effectiveness of forest vegetation in attenuating flows, as indicated by sediment deposition patterns. General recommendations are presented regarding the establishment and maintenance of forested buffer zones along revetted reaches of the Lower Mississippi River.

METHODS

The field studies were conducted in October and November, 1986 in the vicinity of revetments at 23 sites located between River Miles 402 and 857 above Head of Passes, Louisiana. Forested sites subject to high overbank flow velocities were selected for sampling. The sampling design was based on the assumption that deep sand deposits left by the 1973 flood and subsequent high stages would still be evident in forests 15 years or more of age and that the distribution of such deposits would define a zone of substantial flow-velocity reduction. Sand deposited in 1973 or later was distinguished from much older deposits by tree stem burial. Trees rooted on old sands showed evident root flare at the soil surface, whereas those on sites subjected to deposition after tree establishment had straight trunks emerging from the soil surface and no evidence of flaring. Nonforested sites were not sampled because the extent and age of sand deposits could not be determined reliably.

At each sampling location, two or more transects were established at 150 or 300-ft. intervals along the river bank. Additional transects were established where sand deposits were highly variable. Transects were oriented perpendicular to the bank and served as the baseline for plot sampling. At 50-ft. intervals along the transect, stem burial and soil texture (to a depth of 15 in.) were evaluated, and a vegetation plot was established. Broader plot intervals were adopted in stands that were uniform and well represented by previous samples on the same transect. Observation intervals were shortened (by 5-ft. increments) if a sharp change in site or forest conditions occurred (e.g., to isolate the location and influence of a road, swale, or ridge). Transects extended to the edge of deep sand deposits or for at least 200 ft. on sites without significant recent deposits.

Vegetation samples were taken in 6 by 150-ft. plots oriented perpendicular to the transect (parallel to the river). Trees were counted by size class (stem diameters of 2 to 6 in., 6 to 10 in. and >10 in.), and total coverage was estimated for understory (woody plants with stem diameters <2 in.) and ground cover vegetation. Species composition and any apparent effects of sediment deposition were noted in each plot. The field data were evaluated to determine sediment deposition patterns (reflecting flow-velocity attenuation), as influenced by forest structure and floodplain configuration (levee or bluff proximity). These analyses and other published data were used to formulate recommendations concerning the size and management of effective forested buffer strips for flow-velocity reduction along the Lower Mississippi River.

RESULTS

Twenty-three riverbank sites were sampled using 51 transects and 189 vegetation plots. Marked sand deposition was recorded at all but three of the sample sites. Where they occurred, continuous deposits ranged from 100 to 550 ft. from the riverbank, with occasional isolated deposits up to 700 ft. from the river. Average extent was 215 ft., and over 70 percent of the deposits were limited to within 375 ft. of the bank (Figure 1). Deposit depths were estimated to be less than 2 ft. on most sites, although occasional high ridges or filled swales were noted where deposits exceeded 6 ft. in depth.

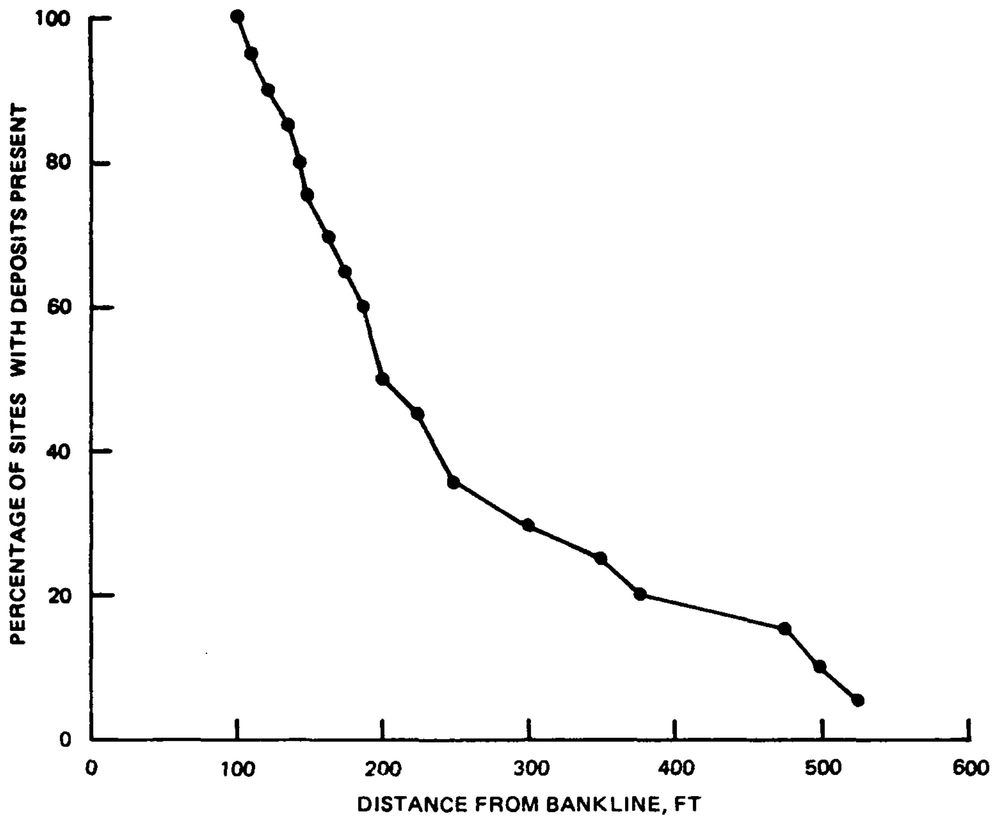

Figure 1. Sediment deposit coverage for 20 sample sites along the Lower Mississippi River. Sites without continuous deposits not included.

Regression analysis indicated no correlation between vegetation structure (at the time of sampling) and sediment deposition patterns. However, in comparison with other forested sites within the Mississippi River levee system (Klimas, 1987), the riverfront forest stands showed markedly lower tree density and moderately higher understory cover. These differences evidently derived largely from the high degree of repeated disturbance (primarily due to shifting road networks) along the riverfront, rather than from sedimentation. Species composition on most of the sample sites was typical of highly disturbed forests elsewhere along the river, with the common occurrence of trees such as sugarberry (Celtis laevigata) and box elder (Acer negundo). Mortality among mature trees was noted in a few instances, but was not particularly characteristic of the sedimented forest stands.

Although no correlation was detected between forest structure and sedimentation patterns, recent forest management and adjacent land uses clearly influenced overbank flow in some areas. The most extensive sand deposits were associated with cottonwood (Populus deltoides) plantations established in the early 1970's, which were probably cultivated rows of very small stems at the time of the 1973 flood. Similarly, stands just downstream of open fields had more extensive deposits than unbroken forest tracts of similar structure.

Floodplain configuration also influenced sand deposition. Deep, continuous deposits did not occur in forests associated with constrictions of the floodplain, that is, where levees or bluffs are within about 1,500 ft. of the river. This was evidently the result of scouring and sediment redistribution, as these same sites often had scattered sand ridges isolated throughout the forested strip.

DISCUSSION

The field survey indicated that where there are well-stocked, structurally diverse forests, overbank flow velocities are attenuated sufficiently to cause deposition of coarse-grained sediments within about 375 feet of the riverbank. On most sites, a zone 50 to 100 ft. wide directly adjacent to the top bank remains partly cleared at all times because of road maintenance. Therefore, an average effective buffer zone consists of about

300 ft. of uneven-aged forest.

A realistic buffer zone recommendation must take into account probable patterns of forest management. Nearly all forests of the areas are subject to regular harvest. Where clearcutting has occurred in recent decades, sand deposits are much more extensive than where selective cutting (removal of particular high-value trees) has been the practice. For example, where stands were clearcut and planted with cottonwood around 1970, the 1973 flood spread deep sands throughout the cleared area and for an additional 250 ft. into the surrounding forest. The even-aged stands that result from clearcutting and/or plantation culture tend to be structurally simple (i.e., they either have minimal tree basal area or minimal understory vegetation) and are therefore likely to be less effective in attenuating flows than are uneven-aged stands.

The arrangement of trees in planted forests can be varied to maximize flow impedence. Shen (1973) conducted a series of simulation studies to determine the influence of tree pattern and harvest system on overbank flow velocities and sediment transport. He concluded that trees established in a staggered pattern relative to the direction of flow greatly reduced flow velocities in comparison with trees in straight rows oriented parallel to the direction of flow. He also showed that the most effective harvest systems for maintaining flow impedence are selective cuts and strip clearcuts oriented perpendicular to the direction of overland flows.

Shen's (1973) simulation studies are applicable to buffer strip management in the Lower Mississippi Valley in several respects. Since most forests of the area are of natural origin, the more effective staggered (random) distribution pattern is common. In the case of tree plantations, both the original planting and subsequent thinnings can be designed to promote maximum flow resistance by maintaining the staggered pattern. Where harvests are planned, both selective cuts and strip cuts oriented perpendicular to flows are almost equally effective in maintaining flow resistance. Currently, selective cutting is a common practice in the study area (Klimas, 1987), but small clearcuts are often the recommended approach to maximize forest productivity (Kennedy and Johnson, 1984). Strip-cutting systems are consistent with the management objectives of small clearcuts and can be used if they are oriented perpendicular to flow direction.

Regardless of the harvest system chosen, the buffer strip must be designed to remain effective at all times, particularly in the years immediately following timber cutting. On the Lower Mississippi River a 300-ft., fully forested buffer is required to slow overbank flow velocities sufficiently to stop sand transport. Assuming most riverfront sites will be subject to harvest, the buffer width must be increased to accommodate the periodic loss of flow resistances. For example, a site that is a candidate for conversion to a cottonwood plantation might be managed as a 600-ft. wide unit, with clearing and conversion of the unit in halves at an interval sufficient to allow trees to reach several inches or more in diameter. Smaller buffer areas may be adequate for certain riverbank zones where current attack is indirect or where extensive timber harvests will not occur. Shen (1973) presents a technique for direct calculation of flow resistance under specific conditions. However, for most revetted sites along the Mississippi River, where overbank flow velocities are often extreme, use of the following general guidelines should improve the effectiveness of forested buffer zones.

1) Buffer zones should extend at least 600 ft. beyond the bankline or beyond any riverfront cleared zone, such as a road.

2) Timber harvests should be limited to removal of no more than half the trees in the buffer zone at any one time, with sufficient time between harvests to allow regenerating trees to reach a diameter of several inches or more.

3) Harvests should be conducted as selective cuts or (preferably) as strip clearcuts oriented perpendicular to the direction of overbank flows.

4) Care should be taken to avoid creating unvegetated corridors through the buffer zone parallel to the direction of overbank flow. For example, roads and rights-of-way should be oriented as nearly perpendicular to flows as possible.

5) Floodplain areas tightly constrained by levees or bluffs may require particular attention with respect to forest management. Field observations indicate that overbank flow velocities are intensified where levees or bluffs occur within about 1,500 ft. of the river.

SUMMARY

Recent forest clearing within the leveed floodplain of the Lower Mississippi River has caused increased overbank flow velocities in some areas (USAED, St. Louis 1985). Erosive flows represent a potential threat to levees and bank protection works as well as to privately developed lands. Forest vegetation has been demonstrated to greatly decrease flow velocities (Shen, 1972). A field study was conducted to determine the minimum forested buffer zone required to significantly slow flows in the high-impact areas associated with revetted banks. The distribution of recent deep sand deposits was adopted as a field indicator of the zone of major flow attenuation.

Where uneven-aged, well-stocked forests were present during floods of the last 15 years, coarse-grained sediments were almost entirely confined to a 300-ft-wide zone adjacent to the river. Sites harvested or cleared during that time had more extensive deposits. Floodplain areas narrowly confined by levees or bluffs showed evidence of turbulent flows even where fully stocked forests were present.

Based on the field research and published simulation studies (Shen, 1973), buffer zone establishment and maintenance guidelines for the study area were devised A 600-ft-minimum buffer zone is recommended, with no more than half of this area being subject to timber harvesting at any one time. The orientation of cleared zones, roads, and rights-of way is of particular concern in the riverfront forest environment.

ACKNOWLEDGEMENTS

The research reported herein, unless otherwise noted, was conducted for the U.S. Army Engineer Division, Lower Mississippi Valley, by the U.S. Army Engineer Waterways Experiment Station. Permission was granted by the Chief of Engineers to publish this information.

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Historical Flooding Patterns and Vegetation Changes Along Shingle Creek, Orlando, Florida

R.W. Ogburn, B.W. Breedlove, and A.M. Watson
Breedlove, Dennis & Associates, Inc.

INTRODUCTION

Shingle Creek is a twenty-mile-long stream that drains 83 square miles of primarily urban watershed in Orange County, Florida. South of Orange County the creek flows through agricultural and urban land in Osceola County, where it discharges into Lake Tohopekaliga. Shingle Creek has been channelized for flood control throughout most of the urbanized area in Orange County. A 0.5 mile segment of the creek between the Florida Turnpike and Sand Lake Road (S.R. 528) has no natural or man-made channel, while it is channelized above and below that segment.

Vegetation near the unchannelized area of Shingle Creek appears to be changing in response to altered flooding patterns. Remnants of slash pine (Pinus elliottii) plantations adjacent to Shingle Creek wetlands are being converted to wetland vegetation, and surface water management systems do not drain as they should based on hydraulic data from channelized portions of Shingle Creek.

A study was conducted to document historical changes in vegetation along Shingle Creek from north of the Florida Turnpike, through the unchannelized floodplain, to the channelized area south of Sand Lake Road, and to relate these changes to changes in flooding patterns. The study included examination of historical aerial photography, as well as field data from upland and wetland areas along channelized and unchannelized portions of Shingle Creek.

MATERIALS AND METHODS

Wetlands examined in this study included a Cypress Dome (Study Dome) north of the Florida Turnpike and floodplain wetlands close to the intersection of Sand Lake Road and Shingle Creek (Figure 1). Planted pines adjacent to those wetlands were also investigated.

Two transects (T1 and T2) were established through planted pines east of the Study Dome and into the wetland. The locations of individual trees and stumps of dead pines were recorded as well as tree height, diameter at breast height (DBH), understory vegetation associations, and water elevations. Selected pines were cored for aging and measurement of growth rates. Ground surface elevation was surveyed at five foot intervals along each transect, and additional key features such as pine stumps and changes in understory vegetation were surveyed. Similar information was collected along transect T3, which passed through planted pines at the eastern edge of the floodplain wetland between the Florida Turnpike and Sand Lake Road. Transect T4 was established in the westside of the Study Dome to quantify understory species composition.

Two wetland transects were established adjacent to the channelized portion of Shingle Creek south of Sand Lake Road. Transect T5 was located west of Shingle Creek, and T6 was east of the creek. Survey information and ecological conditions were documented as on Transects T1 through T4.

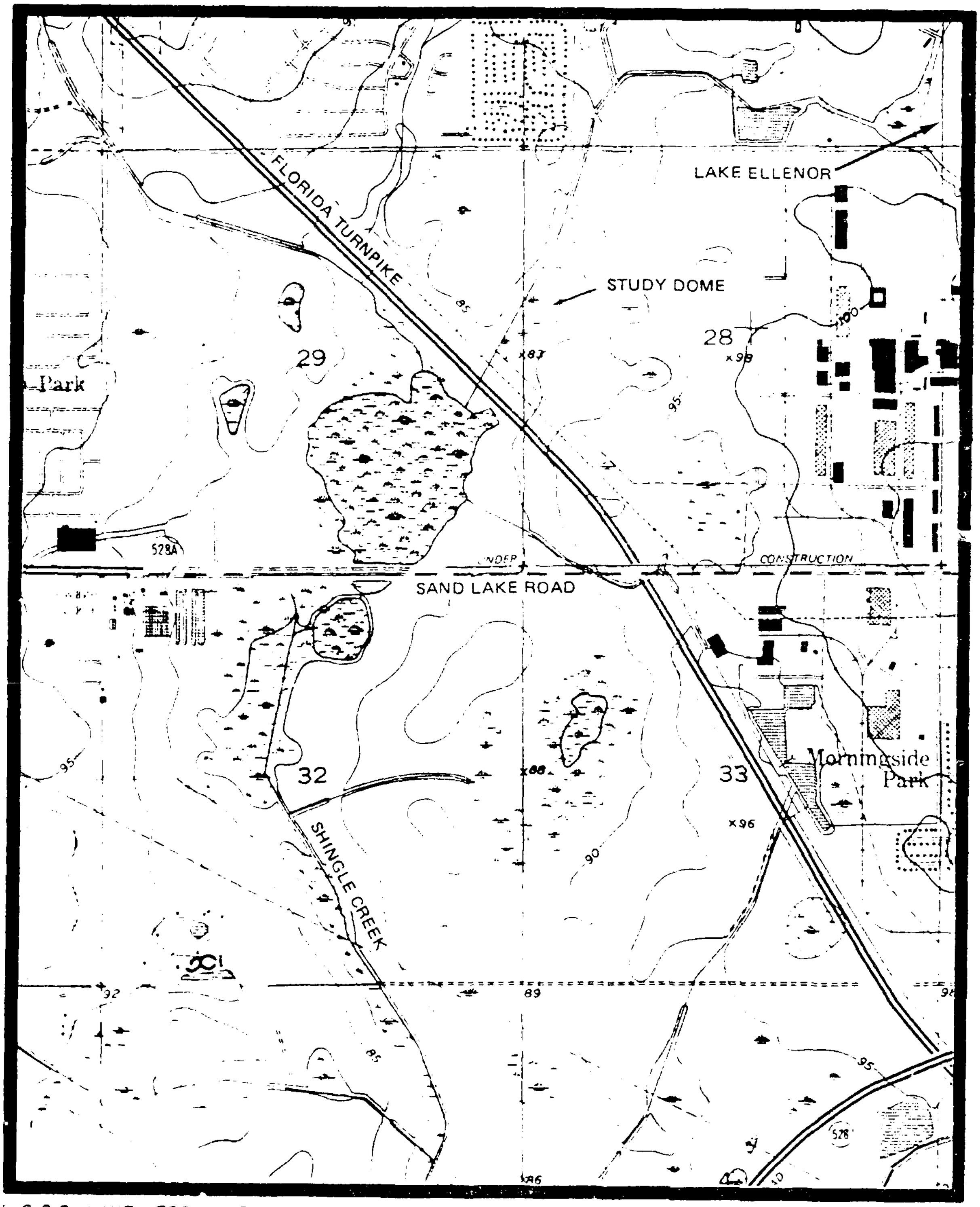
Three 10 foot by 100 foot quadrats (A, B and C) were established in planted pines west of the Study Dome. Percent cover provided by canopy, subcanopy and understory vegetation species was recorded, and elevations at the corners of each quadrat and the mid-points of the 100 foot sides were surveyed. Surface water elevations were surveyed at several stations along Shingle Creek from north of the Florida Turnpike to a point south of Sand Lake Road on April 30, 1985 and May 17, 1985.

A series of aerial photographs of the study area was obtained from the Orange County Engineering Department. They were used to date changes in the study area, such as construction of the Florida Turnpike, widening of Sand Lake Road, and planting of slash pines. They were also used to map changes in wetland and planted pine areas.

RESULTS AND DISCUSSION

Planted Pines

Based on aerial photographs, the slash pines in the study area were all planted around 1963. Data from planted pines near the Study Dome (Figures 2 and 3) and between the Turnpike and Sand Lake Road clearly show the effect of recent increases in flooding. Although the trees are all the same age (about 22 years), those nearest the



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1953 PHOTOREVISED 1980

BREEDLOVE, DENNIS AND ASSOCIATES, INC.
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SCALE: 1" = 2000'

FIGURE 1. SHINGLE CREEK STUDY AREA, ORANGE COUNTY, FLORIDA.

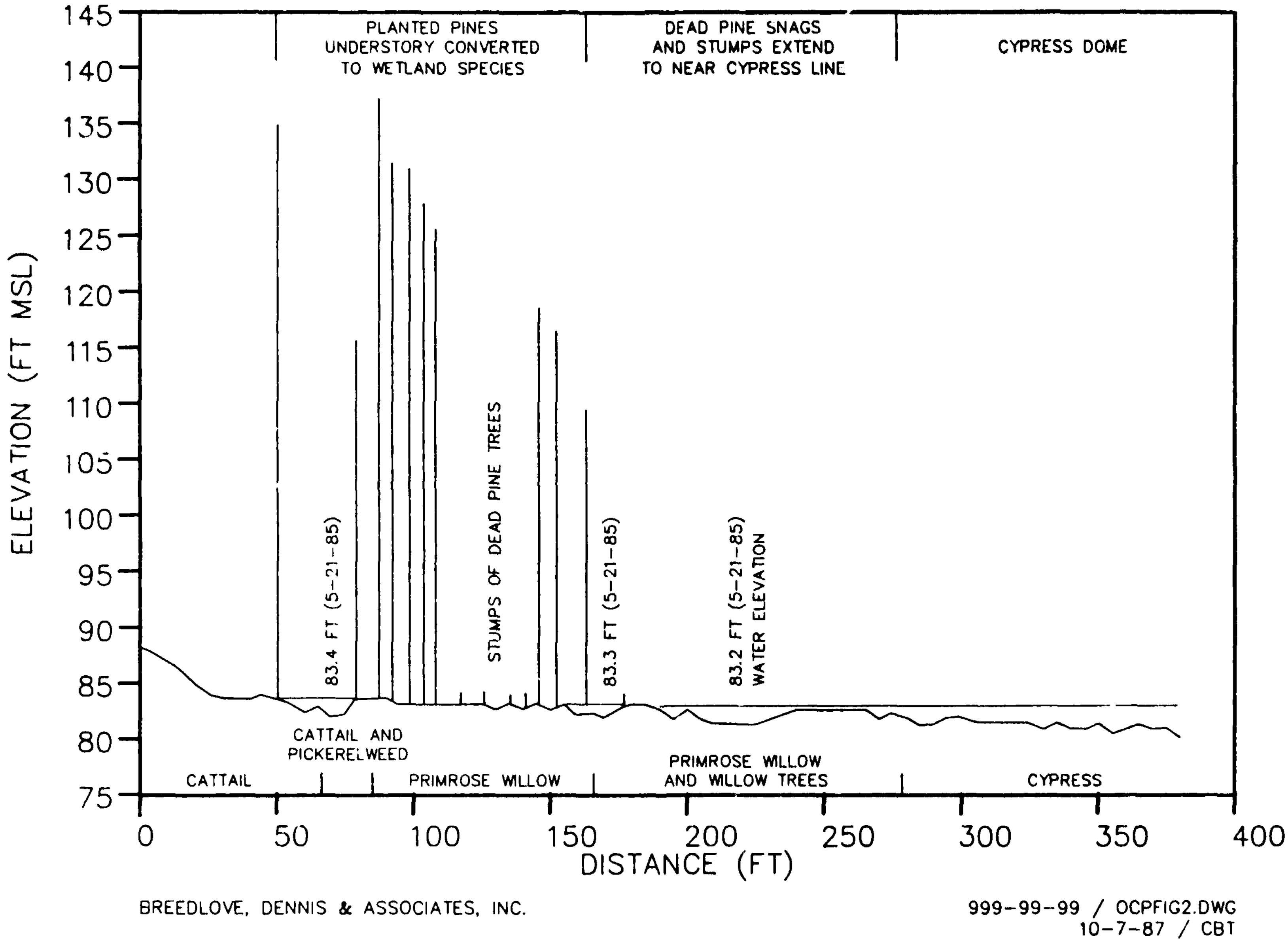
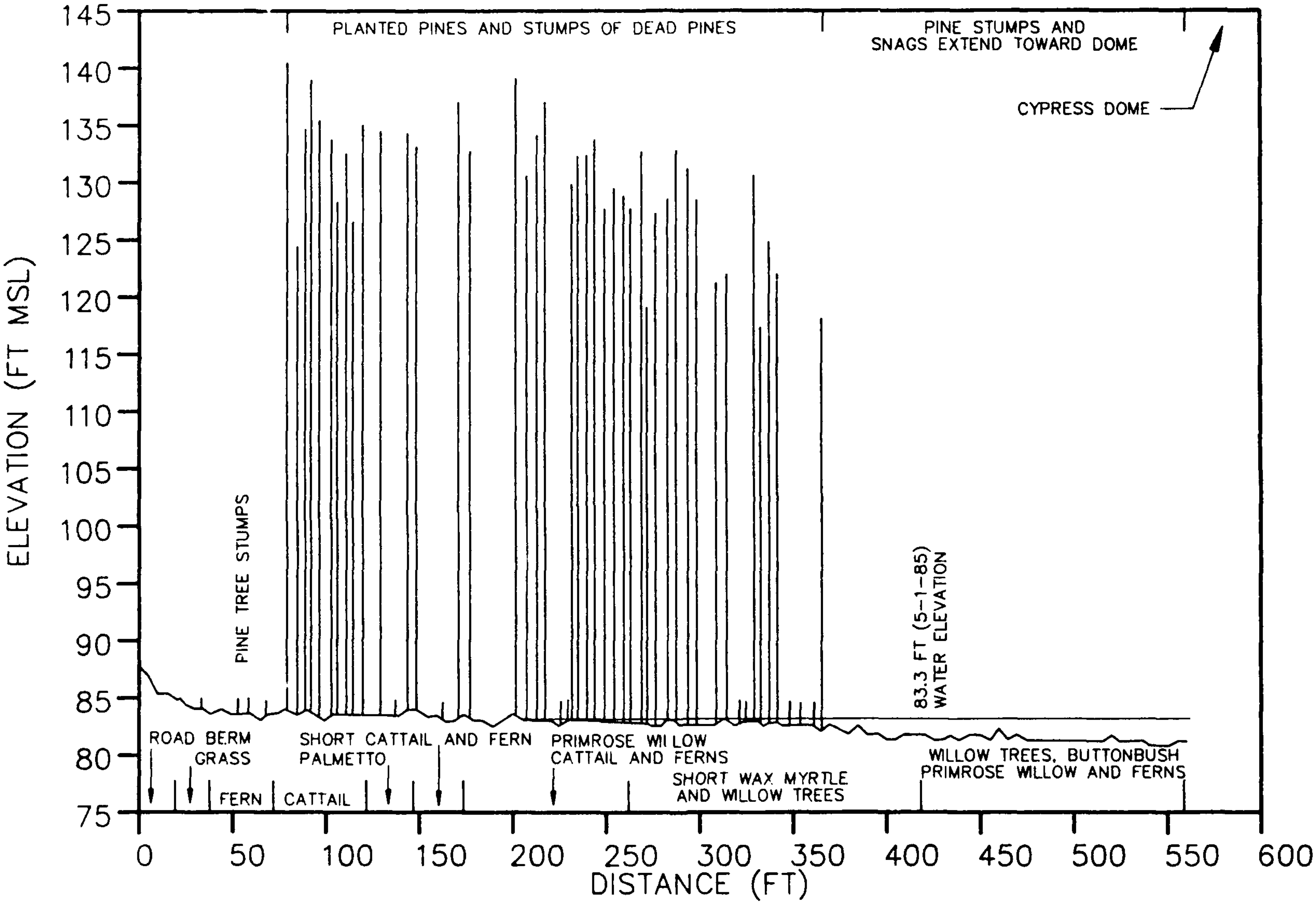


FIGURE 2. PROFILE OF TRANSECT T1 SHOWING HEIGHT OF PLANTED PINES, WATER ELEVATION AND UNDERSTORY VEGETATION.



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999-99-99 / OCPFIG3.DWG
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FIGURE 3. PROFILE OF TRANSECT T2 SHOWING HEIGHT OF PLANTED PINES, WATER ELEVATION, AND UNDERSTORY VEGETATION.

dome and at a lower elevation are stunted or dead on all three transects. Stumps show that planted pines originally grew much closer to the wetlands. A soft muck or silt layer near the water surface that is supported by water roots of primrose willows and willow trees extends from the edge of the living pines into the wetlands and a natural sandy soil lies 2-3 feet below the organic surface layer. Wetlands and pines south of Sand Lake Road do not have similar organic layers.

Understory vegetation in the pines along Transects T1 (Figure 2) and T2 (Figure 3) is dominated by wetland species such as primrose willow, cattail, and pickerelweed, with some palmetto at higher elevations. Scattered cypress seedlings are invading the pines in the Transect T2 area, but no cypress seedlings occur in the Study Dome or in the organic muck area between the dome and the planted pines. Since cypress seeds require saturated but unflooded soil to germinate, it appears that recent constant inundation in the Study Dome has prevented cypress reproduction in the dome itself and in the flooded area between the dome and the pines. The area with stressed pines was dry enough initially for the pines to grow at normal rates for several years, but recently increased flooding is killing the pines and allowing cypress trees to invade.

The vegetation quadrats (A, B, and C) west of the Study Dome showthe transition from healthy pines with palmetto understory to dead pines with wetland understory species (Table 1). Quadrat A, at the highest average elevation (84.4 ft.), was dominated by slash pine and palmetto. Quadrat B, at an intermediate elevation (84.1 ft.), had reduced coverage by slash pine and palmetto and increased coverage by more water-tolerant species such as groundsel and blackberry. Quadrat C, at the lowest elevation (82.9 ft.), had been converted completely to wetland species such as cattails, primrose willow and floating aquatic plants. No palmettos were observed and only stumps of dead pine trees remained in Quadrat C.

The effect of increased flooding also can be seen in tree core data, by comparing growth rates of healthy pines from high elevations with the growth of stressed pines near the Study Dome. Average diameters of 3 unstressed pines near the northeast end of the Study Dome were plotted by age class against the average diameter of 3 stressed trees from the same age classes at increasingly lower elevations (Figure 4). The straight diagonal line in those graphs represents the plot that would occur if both groups of trees had equal growth rates. Deviations toward the horizontal indicate slower growth in the stressed group.

Initial growth rates were similar for both groups of trees at all elevations. Recent growth rates were slower in the stressed group at all four elevations, and the reduction in growth occurred at successively earlier ages as the elevation of the stressed trees decreased. For example, the break in the curve occurs at age 14 (1979) for trees at elevation 83.6 - 83.8 (A), but it occurs at age 7 (1972) for the trees at 82.4 - 83.1 feet (D). This indicates that chronic flooding has gradually worsened, slowing the growth of pines at higher and higher elevations.

Wetland Areas

Vegetation in the Study Dome is different from that in the wetlands below Sand Lake Road, and the differences indicate differences in flooding patterns. Herbaceous vegetation along T5, south of Sand Lake Road and west of Shingle Creek, is dominated by upland and transitional species such as climbing hempweed and royal fern, with no truly aquatic species present (Table 2). Study Dome herbaceous vegetation is dominated by the transitional swamp fern (which was growing only from organic layers on tree trunks near the water line), and floating aquatic species such as duckweed and floating fern. Camphor trees growing among the cypress trees south of Sand Lake Road indicate that flooding duration is much shorter than in the Study Dome. Wetland soil was saturated but not inundated on June 17, 1985, when the Study Dome contained about two feet of standing water.

Additional evidence of differences in chronic flooding can be seen in the location of water roots and lichen lines on trees in the Study Dome (Figure 5) and in wetlands south of Sand Lake Road (Figure 6). A few small water roots were observed near the soil surface on buttonbush shrubs south of Sand Lake Road, but no water roots were present on trees there. Large, 0.5 inch diameter water roots were sprouting from mature cypress and blackgum trees about 2 to 2.5 feet above the Study Domefloor. Water roots are very unusual on those tree species, and they indicate that a nearly permanent increase in water elevation has occurred since the trees matured.

Water Elevations

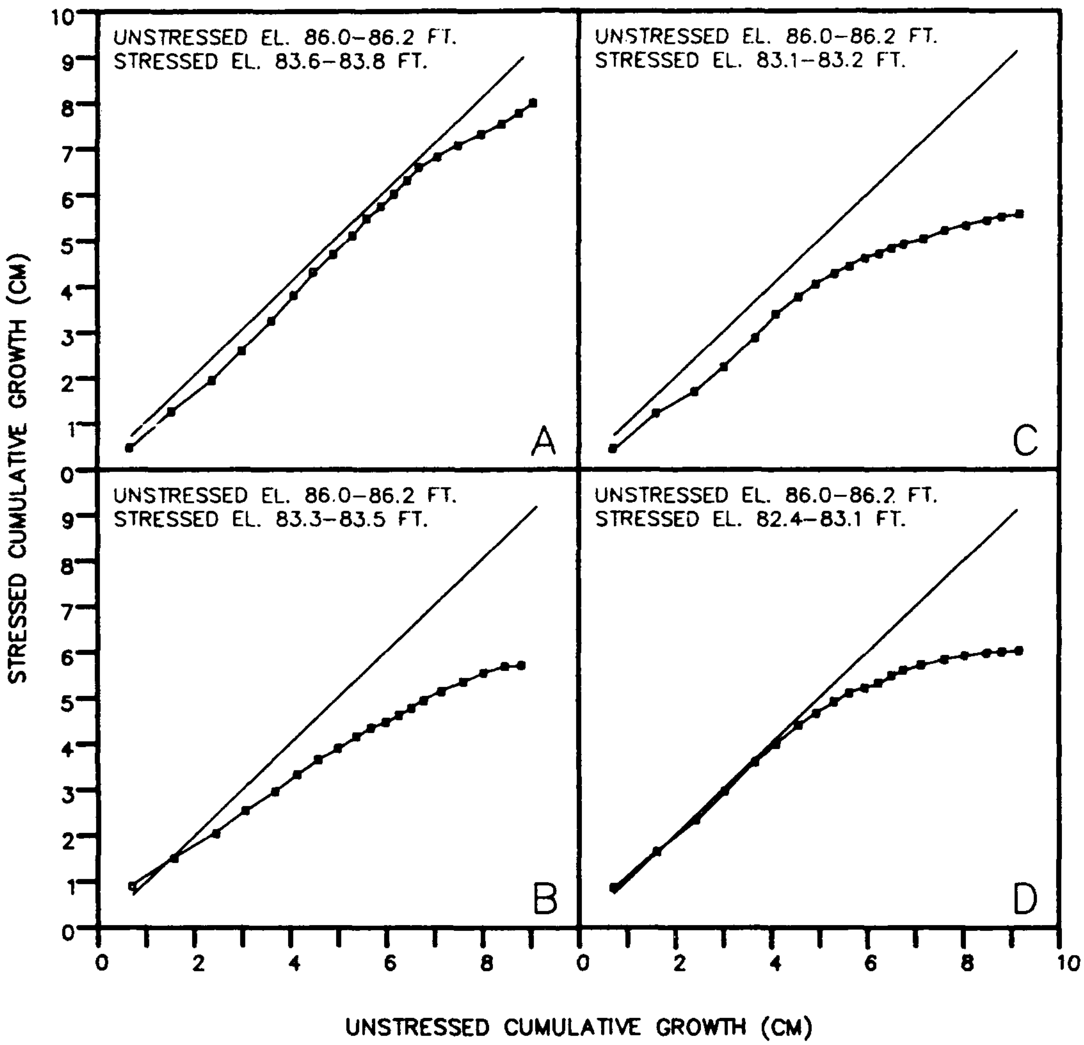
Surveyed water elevations along Shingle Creek from above the Florida Turnpike to below Sand Lake Road show that water is being pooled above the Turnpike (and possibly between the Turnpike and Sand Lake Road), while no pooling is occurring below Sand Lake Road (Figure 7). It is also clear from flooding indicators that the amplitude and duration of flood pulses are different above the Turnpike and below Sand Lake Road. Flooding in the Study Dome shows less variation in depth than in wetlands south of Sand Lake Road. However, flood peaks south of Sand Lake Road do not last long enough to induce the water roots or other indicators of chronic

Table 1. Vegetation Associations in Three Quadrats West of the Study Dome.

	Percent Cover		
	A Healthy Pines	B Transitional Area	C Dead Pines
Canopy			
Slash pine (*Pinus elliottii*)	35	5	0
Subcanopy			
Groundsel (*Baccharis* sp.)	0	1-2	0
Herbaceous			
Saw palmetto (*Serenoa repens*)	100	50	0
Blackberry (*Rubus* sp.)	0	30	0
Primrose willow (*Ludwigia peruviana*)	0	0	80
Cattail (*Typha* sp.)	0	0	10
Pickerelweed (*Pontederia* sp.)	0	0	5
Water hyacinth (*Eichhornia crassipes*)	0	0	5
Water fern (*Azolla* sp.)	0	0	5
Open water	0	0	5
Other forbs	0	20	0
Mean Elevation (Ft. MSL)	84.4	84.1	82.9
Elevation Range (Ft. MSL)	84.0-84.7	83.5-84.7	82.7-83.0

Table 2 Herbaceous Vegetation in the Study Dome and South of Sand Lake Road (Transect T5).

Species	Percent Cover: Study Dome	Percent Cover: Transect T5
Upland		
Climbing hempweed (Mikania sp.)	0.0	20.0
Catbrier (Smilax laurifolia)	0.5	0.0
Poison ivy (Rhus radicans)	0.0	1.0
Transitional		
Red ludwigia (Ludwigia repens)	0.0	8.0
Marsh pennywort (Hydrocotyle sp.)	5.0	0.0
Royal fern (Osmunda regalis)	2.5	37.0
Swamp fern (Blechnum sp.)	15.5	4.0
Wetland		
Water willow (Decodon sp.)	2.0	0.0
Buttonbush (Cephalanthus sp.)	1.0	2.0
Virginia willow (Itea virginica)	0.0	3.0
Aquatic		
Duckweed (Lemna sp.)	35.0	0.0
Mosquito fern (Azolla sp.)	1.0	0.0
Floating fern (Salvinia sp.)	37.0	0.0
Unvegetated	0.0	25.0
TOTAL	100.0	100.0



DIAGONAL LINE REPRESENTS SLOPE FOR NO DIFFERENCE IN GROWTH RATES. EACH POINT REPRESENTS TREE DIAMETER FOR EQUAL AGE CLASS TREES.

BREEDLOVE, DENNIS & ASSOCIATES, INC.

999–99–99 / OCPFIG4.DWG
10–7–87 / CBT

FIGURE 4. COMPARISON OF GROWTH RATES OF STRESSED AND UNSTRESSED PINES AT FOUR DECREASING ELEVATIONS (A–D).

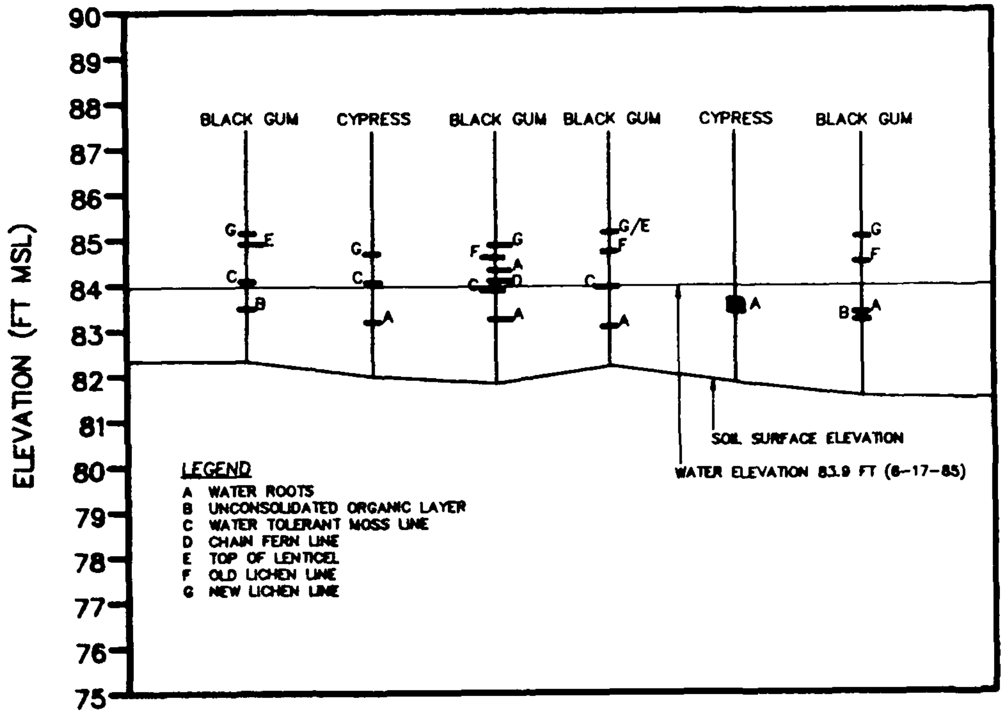
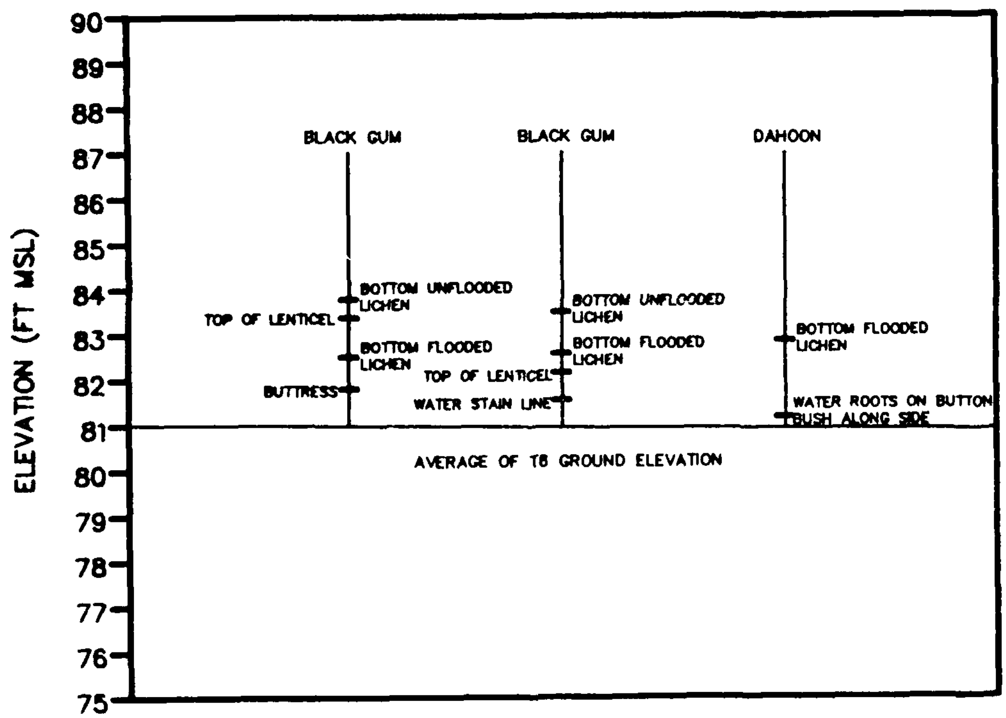


FIGURE 5. FLOODING INDICATORS ON TREES IN STUDY DOME.



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999-99-99 / OCPFIG56.DWG
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FIGURE 6. FLOODING INDICATORS ON TREES AT WETLAND TRANSECT T6, S.E. QUADRANT OF SAND LAKE ROAD BRIDGE.

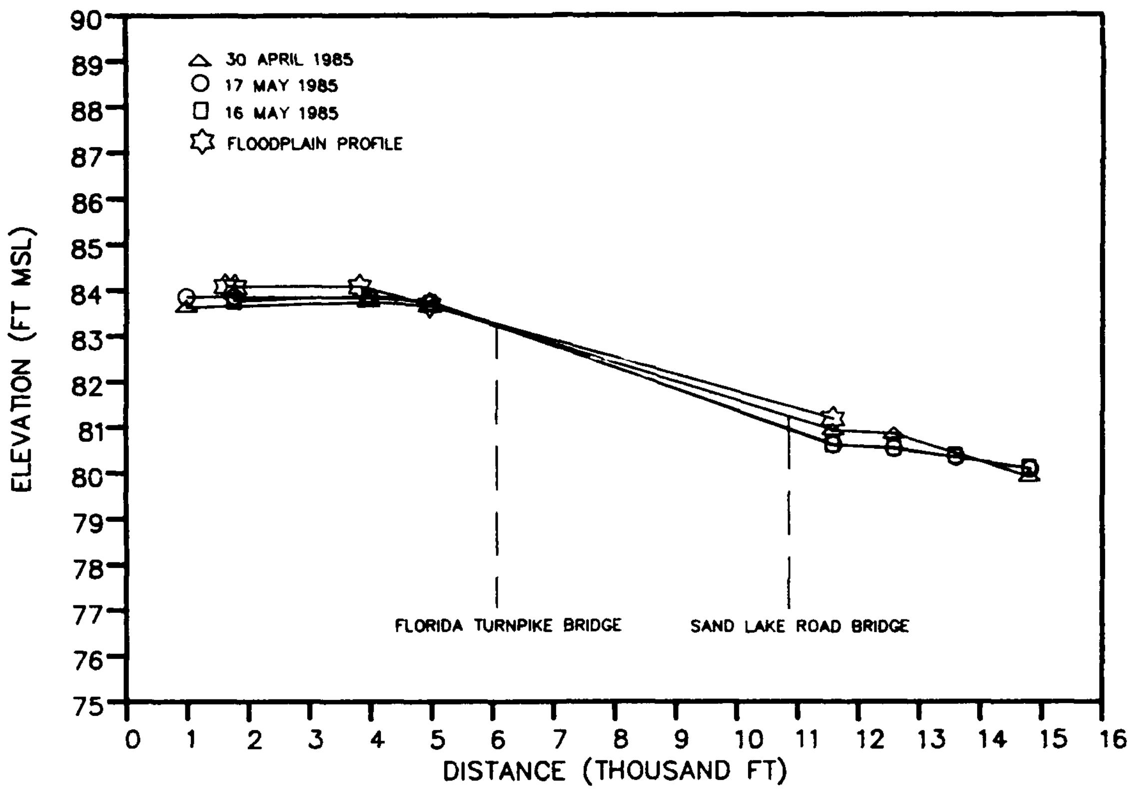

FIGURE 7. SHINGLE CREEK WATER ELEVATIONS IN STUDY AREA.

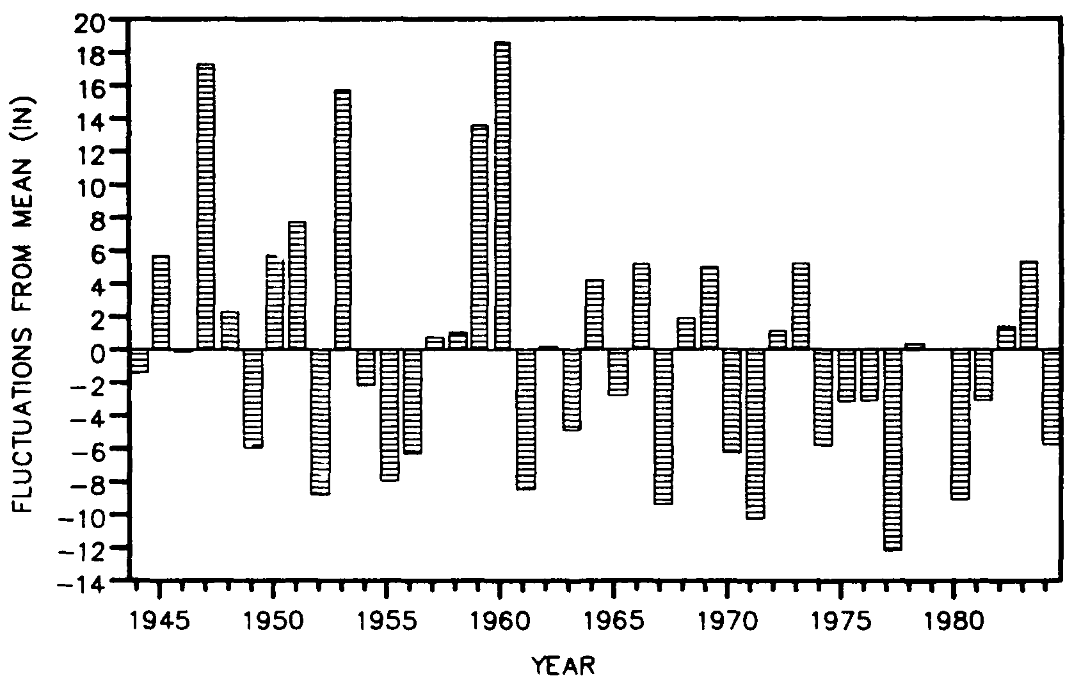

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FIGURE 8. FLUCTUATIONS OF ANNUAL PRECIPITATION RELATIVE TO THE ANNUAL MEAN FOR THE PERIOD 1944-1984 (50.24 INCHES) IN ORLANDO, FLORIDA.

flooding that are seen in the Study Dome.

Long term records from Orlando International Airport show that annual rainfall amounts have consistently been below average since about 1961 (Figure 8). This eliminates the possibility that increases in flooding since the 1960's near Sand Lake Road simply reflect long term variations in rainfall patterns. The rainfall record indicates that flooding should have tended to decrease since about 1961.

CONCLUSIONS

1. Large plantations of slash pines were planted near Shingle Creek wetlands in the vicinity of the Florida Turnpike and Sand Lake Road around 1963.

2. The pines all grew at normal rates for several years, but those at lower elevations north of Sand Lake Road have since been killed or stunted, while the understory vegetation has changed and is continuing to change from palmettos to wetland species.

3. No recent cypress reproduction has occurred in the Study Dome or in the unforested fringe between the dome and the planted pines, although cypress seedlings are presently invading the remaining pines.

4. Flooding indicators such as water roots and lichen lines on wetland trees demonstrate that chronic water levels have increased recently north of Sand Lake Road, but that wetlands along the channelized portion of Shingle Creek south of Sand Lake Road drain quickly.

5. Vegetation patterns and water levels indicate that Shingle Creek flow is being blocked between Sand Lake Road and the Florida Turnpike. This in turn has caused a steady increase in flooding north of Sand Lake Road since the late 1960's, at a time when annual rainfall for Orlando has been consistently below average.

6. The most likely cause of flow blockage is a build-up of sediments onthe floodplain floor in the low-velocity, unchannelized areas.

chapter six

Hydrology and Wetland Quality

Wetland Hydrology and Water Pollution Control Functions

Robert H. Kadlec
The University of Michigan

INTRODUCTION

The purpose of this paper is to describe some less-than-obvious processes involved in the interaction between wetlands and water-borne materials, which include both suspended and dissolved substances. The water budget and water regime of a wetland are known to be the key features to which water quality wetland functions can be connected. The processes of water addition and water removal determine storage status in the wetland as a function of season and environmental factors. The processes of precipitation and evapotranspiration are opposing interactions with atmospheric water. Stream flow in and out of a given wetland ecosystem provides points for ready measurement of incoming and outgoing material. Recharge and discharge phenomena are not visible but do connect the wetland with underlying aquifers. Runoff from surrounding upland areas across the perimeter of the wetland forms another possible input or output for the water pool within the wetland. In some cases the wetland also interfaces with a lake, river or estuary and thus is subject to floods, seiches, and wave action. These interchange processes affect the water chemistry within the wetland and determine the fate of materials which can move with water.

It is important to note that discussion of flows and contents of the water within a wetland must focus upon a defined system. For the purpose of this paper, that system will be taken as the water sheet within the wetland. It is considered separate and distinct from the stationary components of the wetland ecosystem--the soils and vegetation. The seasonal fluctuations of all components of the water budget for a given wetland system are also of great importance. Wet and dry, frozen and unfrozen, and warm and cold seasonal behavior, when coupled with the differences in water regime within the wetland give rise to strong influences on processes involving water-borne substances. These variations influence the vegetative cover, the types and abundance of invertebrates, and the use of the wetland by birds and animals. Most importantly, total inputs and outputs of many materials of interest vary strongly with the above factors. Consequently, samples taken for the determination of water quality are seasonally variable, and total quantities depend on a firm knowledge of the water budget.

It is not enough to simply view the wetland ecosystem as a 'black box'. The internal structure of a particular wetland strongly influences the pathways, timing, and water movement within that system, and thus affects the opportunity for interaction between dissolved and suspended materials.

This paper selects and illustrates four classes of wetland hydrologic processes which may not be apparent under time-constrained observation. First, it is necessary to consider how a material is added to or removed from the water sheet. Second, these localized events combine with water transport to provide a confusing situation at the wetland periphery -- where observations are easiest. Third, many wetlands have dry periods, which cause internal redistributions. Last, northern wetlands experience freezing phenomena, which cause water quality alterations. This list is not intended to be complete, but rather to alert wetland observers to some of the potential influences on water quality.

MASS TRANSFER

The "action" zone in the water column is the interface with solids: roots, stems, litter, or soil. These solid surfaces are the primary locus of bacteria and fungi, which promote chemical alterations such as denitrification or sulfate reduction. Plant and microbial uptake occur in this zone, as do ion exchange and sorption. The intrinsic rates of such processes are often quite rapid. (See, for example, Hammer and Kadlec, 1980; Sapek, 1976; Bartlett, et. al., 1979.) Where there is potential for such surficial conversion, there exists a flux of the chemical through the water to the surface, as shown in Figure 1.

The flux is usually described as proportional to the concentration driving force

$$N = k\,(C - C_S) \qquad (1)$$

If the surface concentration is negligible compared to the bulk, then

$$N = kC \qquad (2)$$

This assumption is valid, for instance, when the concentration in the water is much higher

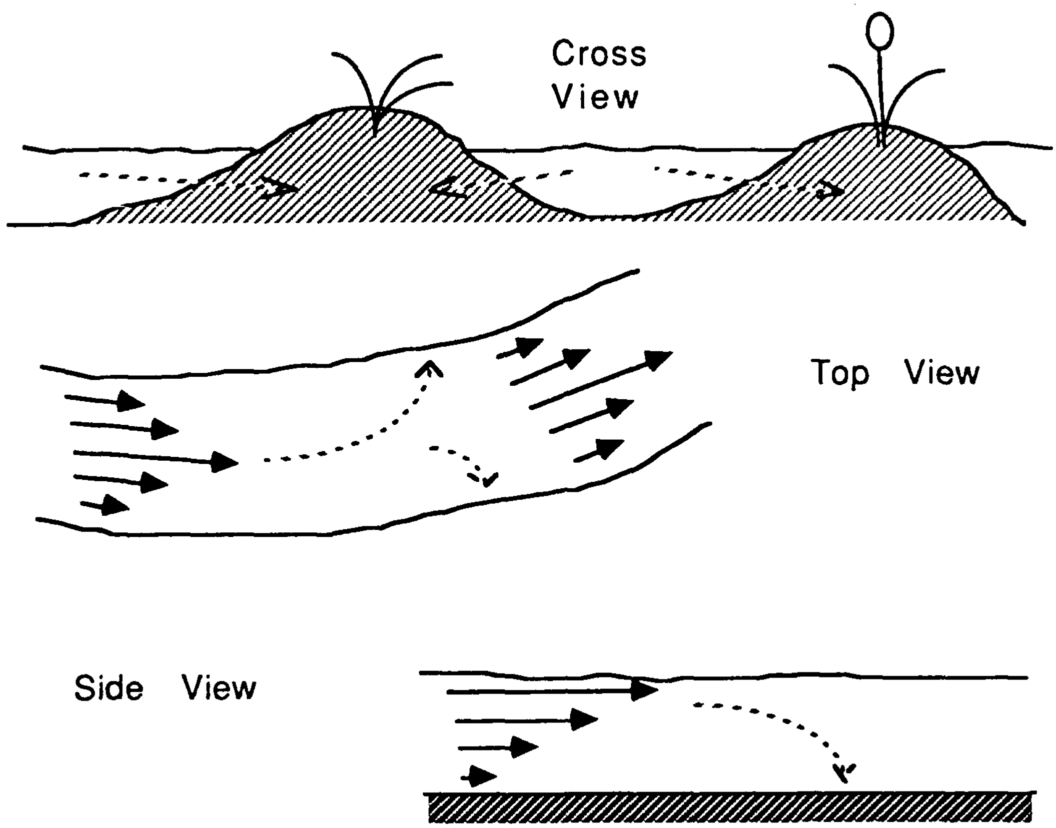

Figure 1. Fluid Flow and mass transfer in wetland channels

than wetland background concentration. Combining Equation (2) with a mass balance for the component in steady one-directional flow gives a logarithmic decrease in bulk concentration with distance:

$$\ln \frac{C}{C_0} = \frac{kZ}{vd} \qquad (3)$$

This model has been found to apply reasonably well in a wide variety of circumstances (see, for example, Tchobanoglous, 1987; or Kadlec and Hammer, 1982). The mass transfer coefficient (k) is a parameter which includes the effects of diffusion and water eddies.

Although (3) embodies some effects of hydrology, namely velocity and depth factors, there are further interactions. The mass transfer coefficient depends on water velocity (Bennett and Myers, 1962):

$$k = b\, v^{1/2} \qquad (4)$$

If there are water control structures, velocity and depth may be independent, but in many wetlands, depth controls flow rate (Kadlec, 1987):

$$v = ad^2S \qquad (5)$$

where the parameter (a) is site-specific. Under those circumstances, (3), (4), and (5) combine to show that concentration reductions are inversely proportional to the square of the depth. Given the somewhat perverse nature of wetlands, the above formulas may not apply precisely. However, the trends are clear: higher flow rates and deeper wetlands mean less chance for interactions. And, in the absence of control structures, deeper water means much greater flow as well as longer component travel to a surface. Figure 2 illustrates these concepts for the Porter Ranch wetland.

Added complications occur in the form of distributions of depth, velocity, and vegetation resistance within the wetland. Phenomena such as channelization are the norm rather than the exception.

TIME AND DISTANCE EFFECTS

Many wetlands collect water from precipitation and runoff from the surrounding watershed, pass it through a complex of

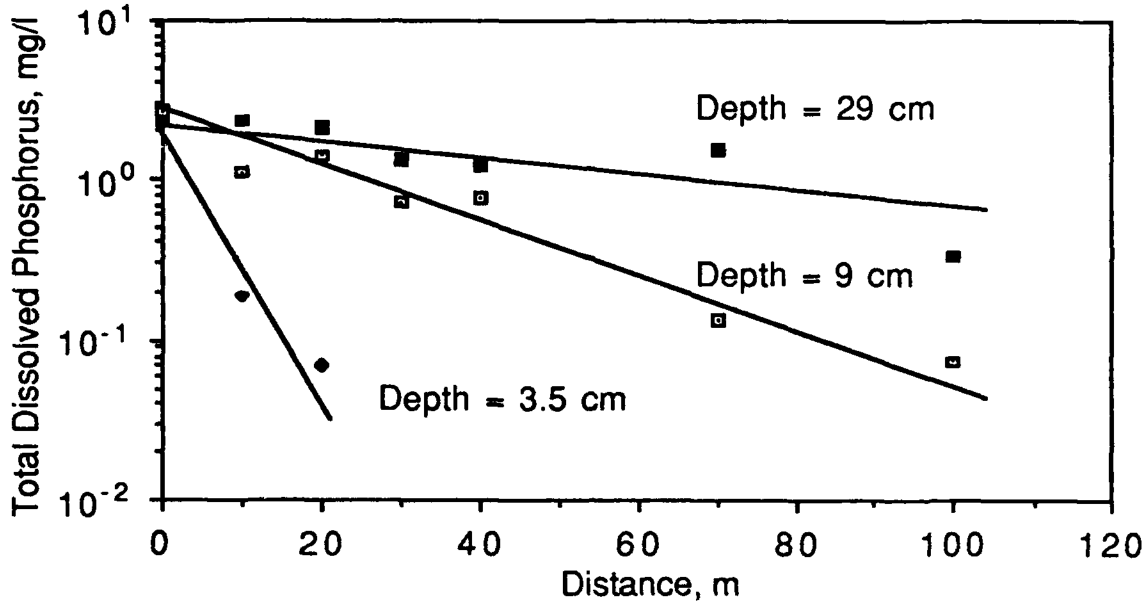

Figure 2. Phosphorus uptake as a function of distance in flow direction. Data from Porter Ranch peatland.

sediments and biota, and discharge it via an outlet stream. Under such circumstances, there are complicated time and distance effects. Input-output measurements at a single point in time are apt to create false impressions, as is illustrated schematically in Figure 3. This is the hypothetical depiction of a wetland system in which seasonal effects of a particular dissolved or suspended substance are present. It is seen that some seasons of the year inputs exceed outputs and consequently accumulations take place and at other seasons of the year the reverse is true, but perhaps not to the same extent. It is thus seen that single input output measurements can lead to erroneous conclusions about long-term storages or removals of the substance in question. For example, during season two, the immediate appearance is that of an exporting wetland yet the long-term trends show that such seasonal export is exceeded in all years by season one import. What is needed to interpret the action of a given wetland is a knowledge of both the input and the output over a significant time period with testing frequency adequate to measure transient events. Single or infrequent synchronized measurements of inputs and outputs have a high probability of producing an erroneous conclusion.

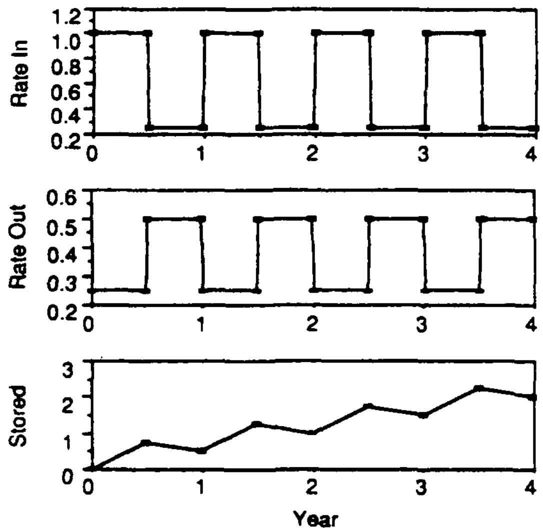

Figure 3. Hypothetical dynamic mass balance components. Inputs and outputs not in synchronization, leading to the possibility of *missinterpretation of wetland function based upon* one-time grab samples.

It is possible that the wetland is not in a stationary state with respect to a particular substance, but is either accumulating or depleting. Two principal portions must be either "filled" or "emptied": the water and the soils plus biota. If there are no interactions with solids, the response to an inlet event at the wetland outlet is delayed by the residence time (turnover time) for the water body. Unfortunately, that turnover time is not easily determined. The nominal turnover time, determined by dividing water volume by flow rate, is frequently a gross overestimate. The existence of a large fraction of deadwater has been found by Hammer and Kadlec, 1986.

The response of the wetland may be delayed further by the chromatographic effect, illustrated

in Figure 4. An elevated chloride input, initiated in 1978, did not appear at the output until 1981. The nominal turnover time for this wetland is known to be about two months, and the actual velocities carry water from inlet to outlet in two weeks. Even this highly mobile tracer ion apparently interacted to some extent with the wetland soils and biota.

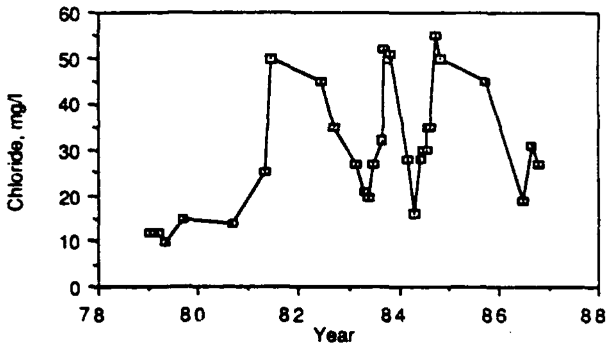

Figure 4. Chloride leaving the Porter Ranch peatland. Additions started in 1978, during the summers only.

A second illustration of the chromatographic effect is given in Figure 5. Phosphorus cycles rapidly to and from soils and biota, which form a very large reservoir compared to the overlying water. This peatland is experiencing a frontal progression in response to repeated phosphorus addition but, after ten years, this front is only a small fraction on the way to the wetland outlet.

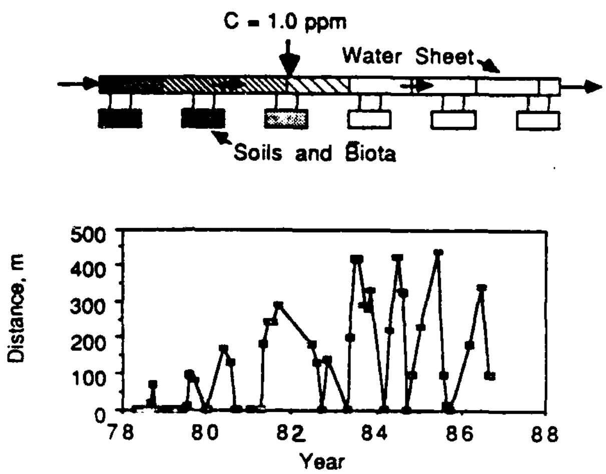

Figure 5. Movement of the point at which phosphorus concentration is 1.0 mg/l. Interactions with the large soil and biomass pools cause this point to move much slower than the water. Data from the Porter Ranch peatland.

DRYOUT

The water regime of many wetlands involves a dry period. The movement of waterborne substances in evaporating flows is significantly different than for steady flows. In one scenario, the inflow to the wetland spreads to form a surface just large enough to evaporate that amount of water. This is the design of the Carson City constructed wetland (Williams, et al, 1987). This scenario also occurs seasonally at other locations (Zoltek, et al, 1979). As water evaporates and moves down-gradient, dissolved substances are swept toward the downstream edge, as shown in Figure 6. This is the reverse of the chromatographic effect.

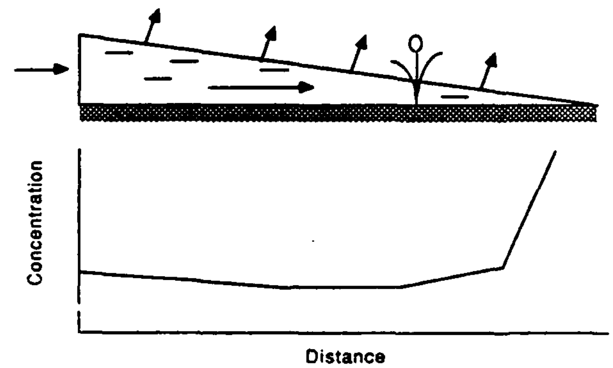

Figure 6. Evaporation causes solutes to become concentrated in the downstream portions of the wetland.

Lateral and vertical solute movements can also occur, as illustrated in Figure 7. Organic soils have a high capillary (suction) pressure, which permits them to remain near saturation at the expense of adjacent water bodies. Thus solutes are moved into the soil pore water and concentrated there by the evaporative processes. In the absence of free surface water, the upper soil layers dry and imbibe water from lower soil horizons. This leads to salt crusts at the soil surface in our western desert wetlands.

These phenomena are predicated on wetland evapotranspiration. The emerging body of information on this subject holds no great surprises, but indicates that there may be deviations from terrestrial behavior. If the wetland exists in dryer surroundings, it may enhance water loss by maintaining full surface moisture potential (Bernatowicz et al, 1976). However, localized shading effects may promote water conservation when compared to upland open water.

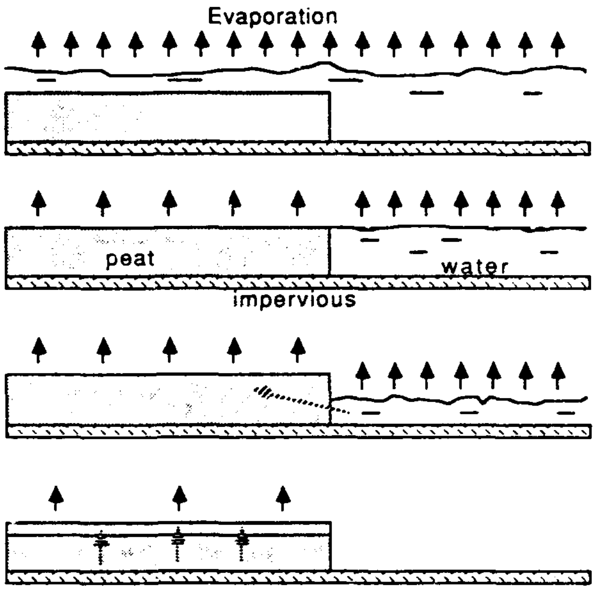

Figure 7. Wicking of water into unsaturated soils moves solutes vertically and laterally.

FREEZING

Most ionic substances are not soluble in ice. Consequently, dissolved materials are rejected from the forming crystalline matrix and concentrate in the underlying water (Li, 1985). Some portion of the solute may be trapped in a pocket within the lattice, so there is an effective distribution between ice and water:

$$C_i / C_W = K \qquad (6)$$

The effective distribution coefficient (K) increases with freezing rate to a value of 1.0 at about 10 cm/hr. For average wetland conditions, K is approximately 0.1 for common species, such as chloride. However, nitrogen and phosphorus are more effectively trapped in the ice, presumably because they are strongly associated with particulate matter. If the freezing front progresses into the soil, solutes are driven into it. There, they are subject to sorption; or at minimum are relocated to immobile interstitial water.

The ice sheet on a flow-through wetland may form on an incline. The water layer below is thus trapped between soil and ice, at the same incline, as shown in Figure 8. The ice sheet is usually imperfect, due to thermally induced cracks, animal activity, or local snow insulation. Underlying water can then flow through a crack, and then downgradient over ice and under snow. A sample of the attendant water quality is given in Table 1.

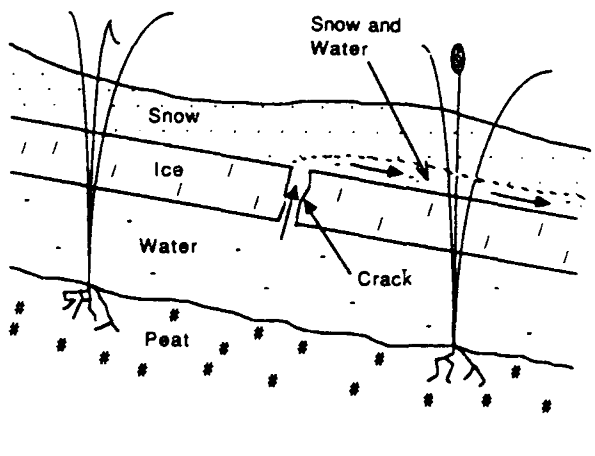

Figure 8. Water can flow up through cracks, and then downgradient.

TABLE 1

Concentrations in the macrostrata of a frozen peatland in central Michigan (Porter Ranch peatland).

	Conductivity, micro mho/cm	Chloride, ppm	pH
Snow (0-50cm)	21	4.6	4.6
Yellow Snow (50-55cm)	390	178	8.2
Over-ice Water (50-55cm)	390	132	7.1
Ice (55-65cm)	229	24	7.0
Under-ice Water (65-70cm)	520	158	6.6

SUMMARY

The transport of waterborne substances through the water to solid surfaces is the first step in their interaction with the ecosystem. This transport is a strong function of both depth and velocity. Apparent velocity is channel dependent. Evaporative processes can rearrange solutes both vertically and horizontally; in some cases preventing their departure. The inputs and outputs from the wetland are by no means in synchronization. It is an error to treat the wetland as a box with an instantaneous response. Ice formation can serve to redistribute the solutes in wetland surface water by driving them downward or upward.

NOTATION

a = constant 1/m·d

b = constant, $(m/d)^{1/2}$

C = concentration, gm/m^3

d = depth, m

k = mass transfer coefficient m/d

K = distribution coefficient,

N = flux, gm/m^{2}d

S = slope,

v = velocity, m/d

Z = distance, m

Subscripts

i = ice

o = inlet

s = surface

w = water

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Nutrient and Sediment Removal in Wetland Buffers

J. W. Gilliam and R. W. Skaggs
North Carolina State University

INTRODUCTION

A very common pattern of rural land use exists in most areas where rainfed agriculture predominates. The agricultural land and homesites are usually restricted to the well drained uplands; native or improved forest are allowed to remain on the steeper side slopes and very poorly drained bottomlands. For example, about 70% of the cultivated land in North Carolina is located in the Coastal Plain. However only 30% of the North Carolina Coastal Plain is agricultural land and most watersheds have at least 50% of their drainage basin in forest (Wilson, 1962). Two watersheds believed to be typical of the North Carolina Coastal Plain were studied by Jacobs and Gilliam (1985a, 1985b). In these watersheds, both surface and subsurface drainage from the agricultural uplands first passes through a series of intermittent streams, flood plains and swamps before reaching a perennial stream (Fig. 1).

Our research over several years on the effects of agricultural drainage on water quality indicated that much of the nutrient and sediment measured in water at the edge of cultivated fields was not leaving the watershed in the streams. Thus, we initiated work to determine the effectiveness of the wetland buffer areas for removing potential pollutants from agricultural drainage water. Research on this topic was also conducted in Georgia (Lowrance et al, 1984a, 1984b) and at the Smithsonian Environmental Research Center in Maryland (Peterjohn and Correll, 1984, 1986).

RESULTS

The effect of a wet riparian area on the nitrate content of subsurface drainage water passing through this zone is shown in Figure 2 taken from Jacobs and Gilliam (1985b). As shown in the upper part of the figure, the hydrologic gradient is from the agricultural field to the stream. There appeared to be an aquatard present at approximately the 3 M depth which prevented deep seepage, so essentially all subsurface flow leaving the agricultural field is believed to be moving toward the outlet at the stream. The seepage area, where the shallow ground water reached the surface before entering the stream, was generally sufficiently reducing, as measured by platinum electrodes, to indicate denitrification would occur. The large decrease in nitrate-nitrogen concentration as compared to change in chloride concentration also indicated that dentrification was occurring. Nitrogen uptake by the riparian vegetation was estimated to be removing only a small percentage of the N from the water passing through the area. The primary removal mechanism was concluded to be denitrification so most of the N was leaving as N_2 gas.

The average nitrate-nitrogen concentration of the small stream (site 4, Fig 1) with agricultural fields on either side containing ground water nitrate-nitrogen concentrations of 10 mgL^{-1} or more was less than 1 mg L^{-1} Thus, approximately 90% of the N was removed as the drainage water passed through the riparian wetland between the fields and intermittent stream. At another location in the same watershed (site 10, Fig. 1) not studied in as much detail, the same removal was apparently achieved with approximately 16 M of riparian wetland. For the watershed, we estimated that an average of 32 kg ha^{-1} yr^{-1} of nitrate-nitrogen was leaving the fields in drainage water. We measured less than 5 kg leaving the watershed in the stream so we believe that 27 kg were removed in the riparian wetlands. Our data indicate that most of the removal occurred in the first few meters of the wetland and the relatively large forested wet bottomland was largely uninvolved with the removal.

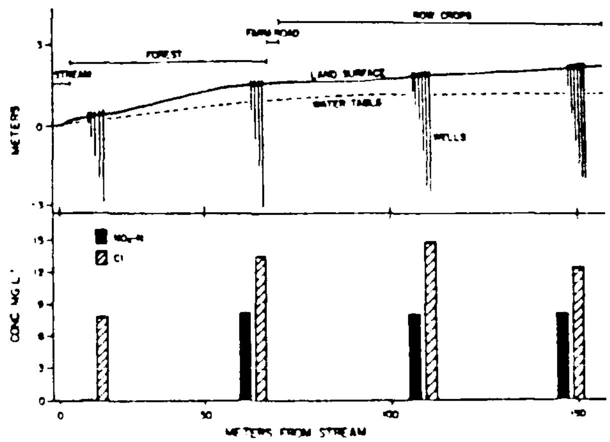

Figure 2. Land use, typical water table elevations, and mean nitrate-nitrogen and chloride concentrations in a Coastal Plain watershed. (From Jacobs and Gilliam, 1985).

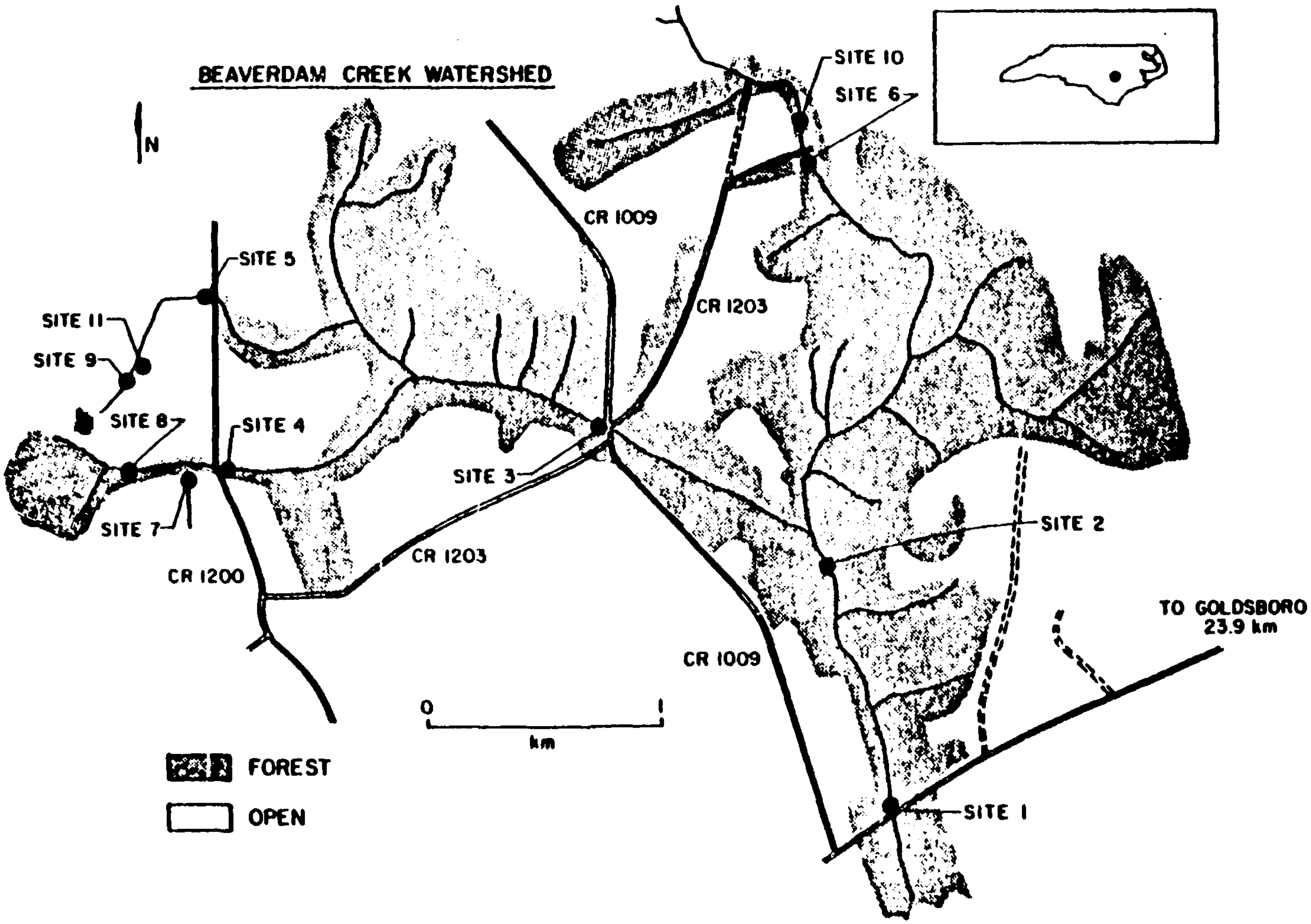

Figure 1. General map of Coastal Plain Watershed studied by Jacobs and Gilliam (1985) showing land use and sampling sites.

In another study, Cooper et al. (1987a, 1987b) used Cesium 137 as a tracer to estimate the effect of riparian wetlands on the deposition of sediment and phosphorus leaving agricultural land in surface runoff in a 1900 ha watershed (Table 1). They found that much of the sediment leaving agricultural fields was deposited in the riparian area very close to the fields' edge. A dense vegetative growth commonly occurs at the field-forest edge. This growth serves to slow the flow of the surface drainage water so that much of the coarser sediment is deposited. A very common landscape feature is sand fans in the edge of the forest where the surface runoff water enters the area.

Table 1. Accumulation of sediment during the past 25 years in various depositional sites in an Atlantic coastal Plain watershed. Data from Cooper et al, 1987a.

Landscape Category	Sediment Depth	Sediment Mass	%
Forest edge	15-50cm	2830Mg	19
1st and 2nd order stream	5-15cm	2840Mf	19
Higher order streams	5-15cm	5900Mg	40
Flood plain swamp	0-5cm	3300Mg	22
Total		14870Mg	100

Frequently the depth of such fans becomes sufficiently great so that the fans form a barrier to water and the flow will be forced to enter the forest at another point, where a new fan will begin to form. As the water moves further down the watershed into the higher order streams and into the flood plain swamp, the texture of the deposited sediments becomes progressively finer. In the floodplain swamp, for example, the sediment layer is much thinner but consists largely of clay sized materials.

The watershed studied by Cooper et al. (1987a) was also used by Humenik et al. (1983) for a 208 Nonpoint Source Best Management Practice study. They measured the loss of sediment and nutrients from the watershed over a three year period. If we assume that the average sediment loss which they measured extended over the previous 25 years when the Cs labeled sediment was deposited, we can estimate the efficiency of the wooded wetland uplands in the removal of sediment. This procedure indicated that 85 to 90% of the sediment leaving the agricultural fields was trapped in the wooded areas.

Cooper et al. (1987b) also measured the phosphorus deposited with the sediment in the wooded wetland during the past 25 years (Table 2). They found that the finer particles deposited in the area of the higher order streams and in the flood plain swamps contained a higher concentration of P than did the coarse materials at the edge of the forest. Even though the flood plain swamp accumulated sediment at a slow rate, the sediments deposited there contained a relatively high concentration of phosphorus. These authors also concluded that the removal of phosphorus by the flood plain swamp required the codeposition of sediment. This conclusion was based upon the observation that the sediments deposited in the swamp could support a higher concentration of phosphorus than was generally present in the

Table 2. The total phosphorus deposited with the sediment in an Atlantic Coastal Plain watershed during the past 25 years. Data from Cooper and Gilliam, 1987b.

Depositional Area	Phosphorus Deposited	%
Forest edge	515kg	6
1st and 2nd order stream	1030kg	13
Higher order streams	3100kg	37
Flood plain swamp	3667kg	44
Total	8312kg	100

stream draining the area. Thus when the swamp was covered with water there would be a tendency for the phosphorus present in the sediments to dissolve into the overlying water. However this process was generally overwhelmed by the deposition of sediment containing phosphorus, resulting in a net deposition. Overall we estimated that the wooded wetlands removed approximately 50% of the phosphorus carried from the agricultural land in surface runoff during the past 25 years.

DISCUSSION

Even though riparian wetlands comprise a large percentage of the rural land area in the Eastern part of the United States, there are very little data on their effect on water quality. In addition to our work discussed here, research in Georgia (Lowrance et al, 1984a, 1984b) and in Maryland (Peterjohn and Correll, 1984) have given qualitatively very similar results. There is a considerable amount of work showing potential nutrient exchange in swamp forests (Brinson, 1977; Kitchens et al, 1975) but our work indicates that the first few meters of the wetland buffers may be much more important than the larger swamps downstream. This is particularity true for the removal of nitrate and sediment from agricultural drainage water.

Although all of our research was conducted on rural watersheds showing the effect of natural vegetation on the removal of sediment and nutrients from agricultural drainage water, there is no apparent reason why the results cannot be applied qualitatively to urban watersheds. Riparian buffer areas between developments and streams should remove much of the contaminants from water passing through these areas.

There is much research needed to answer many of the questions raised by the work thus far completed. We have worked on a very few watersheds. It is likely that other watersheds behave differently with regard to pollutant removal by wetland areas. Some observed but unstudied riparian areas appear to have more distinct channels which may tend to reduce their effectiveness for pollutant removal because of rapid movement (piping) of the runoff water through them. The riparian buffer areas thus far studied have been in native trees and shrubs. A very valid question is "Are planted grass buffer strips just as effective and if so how wide do they need to be to achieve a certain percentage removal?". We also do not know how wide the native buffer areas should be or if type of native vegetation has any influence on the effectiveness of pollutant removal.

We believe that it is imperative that sufficient research be conducted to understand the hydrology and pollutant removal mechanisms so pollutant removal by riparian wetlands can be quantitatively predicted.

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Factors Affecting Wetland Retention of Nutrients, Metals, and Organic Materials

John F. Elder
U.S. Geological Survey

INTRODUCTION

Natural water inflows to wetlands transport nutrients, metals, organic contaminants, and natural organic detritus into the system. Similar constituents are also contained in water flowing out of the system. Because of the widespread interest in the use of wetlands for water quality improvement treatment (for example, Sloey et. al, 1978; Tilton and Kadlec, 1979; van der Valk et al, 1979; Fritz and Helle, 1984; Odum, 1984; Hantzsche, 1985; Willenbring and Weidenbacher, 1985; Mitsch and Gosselink, 1986), it is important to consider how constituent loads change as they are transported through the wetland. The wetland may act as a sink for the constituents, resulting in their net import to the wetland and purification of the water. Conversely, it may act as a source, resulting in net export and an increase in the constituent load of the water. The wetland also may act as a transformer, changing the chemical forms of constituents in the water, without significantly affecting the total loads.

A considerable amount of evidence has been published which confirm that wetlands act as sinks or "traps" for nutrients (Boyt et al, 1977; Spangler et al, 1977; Fetter et al, 1978; Simpson et al, 1978; van der Valk et al, 1979; Kadlec, 1979; Brinson et al, 1984; Dierberg and Brezonik, 1984; Kelly and Harwell, 1985; Day and Kemp, 1985). However, there is also evidence that nutrient retention is highly variable, and, under certain conditions, nutrient release from a wetland may exceed retention (Simpson et al, 1978; Richardson, 1985). Wetland retention capacities are not as well documented for non nutrient aquatic constituents, but the data that are available indicate a great deal of variability in retention capacity.

In short, wetlands can have diverse effects on constituent transport. These effects depend on numerous hydrologic, chemical, and biological processes and the successional stage of the system (Gorham et al, 1979; Meyer and Likens, 1979). Effective management of a wetland for water treatment can be facilitated by understanding the variability and peculiarities of the system.

The purpose of this paper is to summarize current information regarding the factors that can affect the chemical quality of natural and effluent waters during transport through wetlands. Most work in this area has been limited to studies of nitrogen and phosphorus; hence the emphasis of this paper is on those particular elements. However, additional studies of uptake and release effects on metals and organic materials are also included in the review.

FACTORS AFFECTING VARIABILITY OF CONSTITUENT RETENTION BY WETLANDS

Recent studies and reviews of wetland effects on water quality indicate that four general factors--hydrologic characteristics, vegetation, sediments, and microbial activity--are critical to a wetland's capacity to retain constituents in water that flows through the system. Because the relative influence of these factors varies greatly among different wetlands, constituent retention capacities are also highly variable (Hantzsche, 1985; Guntenspergen and Stearns, 1985). Results from one study of a specific wetland can rarely be generalized to others. However, certain principles are frequently reconfirmed in studies of diverse types of wetlands. Some of these general principles are shown in Table 1 as characteristics that favor net retention of constituents within a wetland.

Table 1. Characteristics of hydrology, vegetation, sediments, and microbiota that favor net wetland retention of nutrients and other constituents in flow-through water.

1. Hydrologic characteristics:
   - low slope and low flow velocity
   - outflow characterized by high seepage:drainage ratio
   - long retention time (contact time)
2. Vegetation:
   - high prpoductivity:biomass ratio of vegetation
   - major nutrient input during growing season
3. Sediments:
   - high sorptive capacity of sediments
   - high sediment accretion rates
   - anaerobic conditions in sediments
4. Microbiota:
   - diverse microbial community
   - anaerobic biotransformations

Hydrologic Characteristics--Slope, Velocity, Seepage, Drainage, and Retention Time.

Novitzki (1979) described a system of wetland classification in which four categories are identified, based on hydrologic and structural characteristics (Table 2). Surface-water wetlands are above the level of the water table, and ground-water wetlands have direct contact with it. For each of these types, there are two modes of water outflow - seepage or drainage - depending on whether the structure of the system is a depression or a slope.

Among these are slope, flow characteristics, channelization, flood dispersal, substrate permeability, density of vegetation, and general "roughness" characteristics (features that impede water flow through the system). However, excessive flow retention may be detrimental. For example, Wile et al, (1985) suggested that a 7-day detention time was optimal because longer retention could produce severe anoxia and associated reducing conditions that diminish treatment efficiency. Also, very low surface-flow velocities can diminish transport of particulate organic matter that serves as a base for downstream food webs (Mattraw and Elder, 1984; Livingston et al, 1975).

Table 2. Some distinguishing features of hydrologic wetland types (adapted from Novitzki, 1979).

	Surface Water		Ground Water	
	Depression	Slope	Depression	Slope
Intercepts water table?	no	no	yes	yes
* Outflow	seepage	drainage	seepage	drainage
* Inflow		flooding of adjacent lake or river	GW discharge	GW discharge
Common physiographic associations	basins, ponds, marshes	margins of lakes or rivers	basins, fens, marshes	springs, stream headwaters

Seepage outflow is generally slower than surface drainage, allowing more contact time during which constituents may be removed from the water. Contact with the substrate is especially significant because sorption to sediments is more effective than other mechanisms for removal of toxicants (Nessel and Bayley, 1984; Dierberg and Brezonik, 1984). It has been documented that sediment retention is generally greater in depression (seepage) wetlands than in slope (drainage) wetlands (Novitzki, 1979), but few data are available to verify that the same principle applies to dissolved constituents. Nutrient retention may be significant even in systems where surface-water drainage is apparently a major outflow route (Mitsch et al, 1979; Delaune, Reddy, and Patrick, 1981).

Contact time between flow-through water and the wetland soil and vegetation is particularly important. A minimum contact time (on the order of several days) is critical for the wetland filtering function to be effective (Brinson, 1985; Hantzsche, 1985; Wile et al, 1985). Hence, all factors that control contact time are especially significant.

There are vast differences among wetlands in their retention times and the duration and timing of their flushing cycles. These factors are influenced by physiographic features, drainage, climate, vegetation, and sediments (Brinson, 1985). Some systems, such as interior bogs, may remain relatively isolated from exchange with adjacent aquatic systems for several months. Others, such as tidal swamps, seldom retain water for more than a few days.

Vegetation -- Nutrient Uptake and Organic Production Rates.

Nutrients tend to be rapidly consumed by wetland vegetation to support primary productivity (Mitsch et al, 1979; Whittaker et al, 1979). Vegetation is also a major source of organic detritus that can be transported to downstream ecosystems (Simpson et al, 1978; Elder and Cairns, 1982). This dual role of wetland vegetation results in the transformation of nutrients observed in some wetland systems. As water flows through

the system, the composition of its nutrient loads tends to shift from a predominance of forms used by vegetation to a predominance of forms released by vegetation. Nutrient-uptake rates are highly dependent on the type and density of vegetation. High stem density also increases the resistance to flow, thereby increasing water contact time.

An example of the transformation effect is the Apalachicola River-wetland system in northern Florida. Mass balance studies there by Mattraw and Elder (1984) showed that the annual outflows of water and total nutrients (nitrogen and phosphorus) at the river mouth were within 20 percent of annual inflows at the headwaters of the river. However, greater changes were observed for certain nutrient fractions (Elder, 1985). Bioavailable nutrient forms, including ammonia nitrogen, orthophosphate phosphorus, and, to a lesser extent, nitrate and nitrite nitrogen, were incorporated biologically, whereas particulate, organic forms were released and transported out of the system.

Because nutrient demand is greatest when growth and primary productivity rates are highest, nutrient removal by a wetland will very likely be accelerated during growing seasons. Therefore, if maximum nutrient inputs to the wetland coincide with the growing season vegetative nutrient uptake should be maximized (Simpson et al., 1978; Richardson and Nichols, 1985). In addition, systems that are in a successional state -- that is, where the productivity:biomass (P:B) ratio is high -- are more likely to incorporate large quantities of nutrients than mature systems where the P:B ratio is low (Odum, 1969).

Toxicants are not consumed by active uptake in plants, although vegetative uptake does occur for at least some metals (Giblin, 1985). Metal uptake in plants is partly attributable to sorption to the organic surfaces rather than to active uptake.

Sediments -- Adsorption, Precipitation, and Accretion.

Sediments are critical to wetland retention of dissolved and suspended constituents. These constituents may be fixed by adsorption or precipitation processes, then deposited permanently or temporarily in the sediments. Metals and organic compounds readily adsorb to charged surfaces of clays, silts, and organic particles (Dossis and Warren, 1980; Pavlou, 1980). Metals also are precipitated (Hem, 1985) either as insoluble metal sulfides in reducing environments (Giblin, 1985) or as insoluble metal hydroxides in oxygenated waters with high pH.

Constituent transfer to the sediments, however, does not necessarily result in permanent removal. Both adsorption and precipitation are reversible processes, and they are sensitive to changes in physical and chemical conditions. For example, a decrease in pH may redissolve metal hydroxide precipitates and promote desorption of metals from mineral and organic surfaces (Boto and Patrick, 1979; Hem, 1985). Increases in flow velocity may cause resuspension of recently deposited sediments and their associated contaminants (Turk and Troutman, 1981). Macrophytes incorporate nutrients from the sediments into their root systems, thereby mobilizing sediment-bound nutrients (Prentki et al, 1978; Klopatek, 1978). This "nutrient-pump" process can be an important source of internal loading of nutrients, particularly phosphorus.

Accretion, or burial of old sediments as new material is deposited can lead to permanent transfer of constituents to the sediments (Mitsch and Gosselink, 1986). Once the substances are buried at depths beyond the root zone of wetland vegetation, they are no longer readily recycled by biogeochemical processes (Day and Kemp, 1985), and the system functions as a sink for these substances.

Microbial Activity -- Biotransformation of Pollutants.

The action of bacteria and other microbiota is likely to be of major significance in transformation and retention of aquatic constituents in wetlands (Whelan et al, 1976; Odum, 1984). Most research to date has concentrated on relatively few microbially-mediated transformations. Such research has consistently demonstrated the importance of processes such as sulfate reduction (Giblin, 1985) and denitrification (Richardson et al, 1978; Odum, 1984; Day and Kemp, 1985) in removing nutrients and metals from dissolved phases Godsy et al (1983) and Roberts (1987) have demonstrated anaerobic microbial degradation of synthetic organics in other types of systems. Similar processes may be active in wetland soils, which are usually anaerobic under saturated conditions (Mitsch and Gosselink, 1986).

IMPLICATIONS FOR MANAGEMENT

An overview of information available on wetlands and water quality indicates that many types of wetlands have a capability for improving water quality by retaining or transforming nutrients and toxicants. If constituent retention is indeed the management objective for a particular wetland, the retention capability can be maximized by creating conditions that favor net constituent retention (Table 1).

The use of wetlands for treatment of wastewater generally compares favorably, from an economic standpoint, with other water purification alternatives (Sutherland, 1985; Fritz et al, 1984). Exceptions to the cost advantage occur because of

high land costs or remote location of the wetland. In such cases, artificial wetlands may be used to reduce costs and still achieve the favorable characteristics listed in Table 1 (Wile et al, 1985; Brennan, 1985). Perhaps more important than monetary costs are the potential changes in the wetland itself. Some issues to consider in determining wetland management policy are long-term changes, introduction of pathogens, and biotransformation of non-toxic constituents to more toxic forms that may detrimentally affect the ecosystem.

Long-Term Changes.

Application of wastewater to a wetland for a year or more is likely to produce long-term physical or functional changes in the system. One possible long-term change is that the wetland may become saturated with certain constituents (Richardson, 1985; Guntenspergen and Stearns, 1985) and lose some of its retention capacity for those constituents. The wetland may even become a net exporter of the same constituents. Another possibility is development of channels, leading to shorter contact time, thereby reducing the retention of constituents.

A third possible long-term effect of wastewater application to a wetland is the impact on wildlife. Friend (1985) pointed out that wastewater disposal in a wetland may be detrimental to wildlife by: (1) increasing the likelihood of pathogenic disease, (2) introducing pollutants that interfere with general health and defense mechanisms, and (3) changing the physical and chemical environment, thereby reducing availability of suitable habitat, food resources, and shelter. Birds, which are dependent on physical structure as well as species composition of wetland vegetation, may be particularly susceptible to environmental changes (Kadlec, 1985).

If the volume of wastewater is significant relative to the natural water flow through the wetland, the additional input may have appreciable effects on the hydrologic characteristics. Hydrologic changes can, in turn, produce other long-term effects on water quality, vegetation, microbiota, and wildlife. If hydrologic changes are not considered when designing a wetland treatment system, the response and effectiveness of the system may be substantially different than expected.

There are several options available for practices that may be implemented to control long-term hydrologic effects of wastewater application (Hantzsche, 1985; Richardson and Nichols, 1985). Wastewater inflow may be routed and dispersed in a way that minimizes channelization and maximizes effective contact time with the wetland sediments and vegetation. It may be held in reservoirs and released into the wetland during growing seasons when uptake by biota will be optimal, thereby minimizing the possibility of constituent saturation.

It is possible that proper control of the character and dose rate of inflow may diminish the long-term effects on wildlife, species composition of vegetation, and physical structure. The effectiveness of this manipulation will depend on availability of background information to define the sensitivity of the wetland to increased inflows.

Introduction of Pathogens

Wastewater inputs to wetlands are likely to carry substantial amounts of bacterial and viral pathogens, posing a health hazard to humans and local wildlife. Viruses, in particular, tend to be persistent in aquatic environments (Shiaris, 1985). Grimes (1985) cautioned that a wetland used as a treatment system can become a "disease reservoir". Pathogens in wetlands pose a greater hazard than pathogens in open water systems because of the lack of currents or wave action that could dilute or dissipate the organisms. This presents a management dilemma; high flow velocities might alleviate the pathogen problem but would reduce the system's capability for retention of other constituents (Table 1).

Pretreatment of wastewater may reduce the pathogen load before the water is released to the wetland. If chlorination is used, however, there is a potential for formation of chlorinated hydrocarbons in the wetland system by combination of chlorine with natural organics (Richardson and Nichols, 1985). Elevated chlorine concentrations may also be toxic to natural microbial populations that are responsible for biodegradation of toxic substances in the wetland sediments. For these reasons, Richardson and Nichols (1985) suggested ozonation as a preferred pretreatment process.

Biotransformation of Non-Toxic to Toxic Forms.

Microbial action in mediating chemical transformation of contaminants has a critical role in water quality improvement within wetlands. However, microbial activity can also have unfavorable effects. Although biodegradation may detoxify some toxic agents introduced to a wetland system, it may have the opposite effect on others (Shiaris, 1985). For example, microbiota may transform amines and methyl amines, leading to production of the carcinogen dimethylnitrosamine (Ayanabe and Alexander, 1974). Methylation of metals produces compounds that are likely to be more mobile and more toxic than inorganic compounds of the metals (Jernelev and Martin, 1975; Baker et al, 1983). The possibility that certain toxicant problems may be aggravated, rather than diminished, by microbial action needs to be recognized by designers or managers of wetland treatment systems. If, in specific cases, biotransformation is observed to be a real problem,

pre-treatment procedures may be required to eliminate the precursors of the newly-formed contaminants.

SUMMARY

Many types of wetlands can be useful for improvement of water quality, as has been demonstrated by numerous studies. However, wetlands are complex systems, and the effectiveness and risks of water treatment are highly variable at different sites. Processes that affect water quality in one wetland are not necessarily predictive of processes in another, even if the two systems have similar characteristics.

A wetland seldom functions as a true sink for nutrients and other contaminants. It is more likely to have a multiple role as a source, sink, and transformer, depending on location, season, and environmental factors. Hydrologic features, sediments, vegetation, and microbial activity all influence the wetland's ability to retain constituents contained in inflow water while simultaneously releasing organic matter and other substances that may be valuable for downstream ecosystems. Long-term changes are possible where wastewaters are introduced to wetlands for treatment, and the changes may be detrimental not only to the general health of the ecosystem but also to its effectiveness as a treatment system.

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Rapid Assessment of the Hydraulics and Hydrology of Wetlands for Wastewater Treatment

Martha Jarosewich
WAPORA, Inc., Environmental Consultants

INTRODUCTION

Hydrologic and hydraulic characteristics of freshwater wetlands must be considered in evaluating the use of wetlands for wastewater management. This paper, a summary of the hydrology and hydraulic analyses presented in EPA's Freshwater Wetlands for Wastewater Management Environmental Assessment Handbook (1985), reviews the methodology of the Basic Analysis developed to evaluate and estimate water flows, velocities, depth residence times and areas-of-inundation in wetlands under natural conditions and after wastewater application. The method utilizes fundamental hydraulic principles to conduct a quick, systematic desk-top assessment of the hydraulic character of a wetland and the magnitude of the effects of wastewater discharge to a wetland, while optimizing the use of regional and local information and minimizing field work.

METHODOLOGY

The hydrology and hydraulic analysis methodology includes three levels of analysis: a Basic Analysis; a Seasonal Analysis; and a Refined Analysis. The Basic Analysis is used as an initial screening procedure. The Seasonal Analysis is used when wastewater is to be applied at varying rates through the year or when seasonal variability in hydrology and climate are known to occur in the wetland. The Refined Analysis is used for unique or sensitive wetlands or when basic and/or seasonal analysis indicate the potential for large changes in wetland hydrology due to wastewater application. Each of the three methods can be applied to closed hydrologic wetland systems (e.g. cypress domes, pocosins); open hydrologic systems with an identifiable channel (e.g. bottomland hardwood swamp); open systems with no identifiable stream channel (e.g., marsh cypress stands); and open hydrologic systems with controlled outflow.

A Basic Analysis is performed to estimate the changes in annual average wetland hydrologic characteristics based on published data available on climatology, topography, and geohydrology and site-specific data obtained from a one-day survey of the wetland. The survey would include identification of channel width and bank height, vegetation distribution in the wetland, and a hand-level determination of the elevation change across the wetland perpendicular to the general slope of the wetland.

The Seasonal Analysis is performed to estimate changes in wetland hydrologic characteristics based on seasonal data. Seasonal analysis methods are the same as those of the Basic Analysis, with the exception that the Seasonal Analysis requires monthly data from published sources in addition to the site-specific data obtained for the Basic Analysis.

A Refined Analysis should be performed if the proposed wetland system is unique or sensitive, or if an evaluation of the Basic or Seasonal hydrologic analyses indicates that the wetland would be significantly affected by the wastewater application. A Refined Analysis also should be performed if the hydraulic characteristics are unsuitable for the necessary removal of wastewater pollutants. Data collection should include at least one year of measurements of surface water inflows and outflows, precipitation, evapotranspiration, water surface elevations, groundwater elevations at several locations, and flow path and velocity measurements using tracer studies at various locations in the wetland.

Depth, velocity, and area-of-inundation data collected for the Refined Analysis are compared with predictions made using the Basic or Seasonal Analysis methodologies. Inputs to these analyses are adjusted to reproduce observed field data under existing conditions. These analyses are then performed for conditions present under the application of wastewater to the wetland.

The Basic Analysis

The Basic Analysis is a useful preliminary screening tool to identify situations in which the application of wastewater could cause major changes in wetland hydrology. The analysis is performed in three steps (Figure 1). The first step considers the wetland in its current state, that is, unaltered by any wastewater application. The second step considers the wetland hydrology and hydraulics with the application of a known wastewater volume. The third step compares

hydrologic and hydraulic characteristics of flows in the wetland prior to and with the wastewater application. Depending on the magnitude of the changes, a similar procedure is conducted with a Seasonal or Refined Analysis.

Steps one and two are conducted in two parts. First, a water budget is calculated for the wetland to determine water inflows and outflows. Second, depths of flow, velocities, areas of inundation, and residence times are estimated using Manning's equation.

BASIC ANALYSIS

EXISTING CONDITIONS
(STEP 1)

WITH WASTEWATER APPLICATION
(STEP 2)

WATER BUDGET ANALYSIS
(PART 1)

WATER BUDGET ANALYSIS
(PART 1

MANNING'S EQUATION ANALYSIS
(PART 2)

MANNING'S EQUATION ANALYSIS
(PART 2)

HYDROLOGIC CHANGE ANALYSIS
(PART 3)

ADDITIONAL HYDROLOGIC/
HYDRAULIC ANALYSIS
REQUIRED

SEASONAL ANALYSIS

REFINED ANALYSIS

Figure 1. Flow chart for a basic analysis.

Water Budget Analysis

The water budget analysis is performed to estimate surface water flows in the wetland. The water budget equation denotes the change in water volume stored in the wetland over a specified time period or the difference between water volume inflows and outflows in the wetland. For a specified time interval (t), the water budget equation may be written as:

$$\Delta S = P + Q1 + QL + G1 + W - Q2 - E$$

where:

t = specified time interval over which water budget is calculated

ΔS = volume change of water stored in the wetland

P = precipitation volume falling on the wetland

Q1 = surface water volume flowing onto the wetland at its upstream end

QL = lateral overland flow volume flowing into the wetland

G1 = groundwater volume flowing into the

W = wastewater volume applied

Q2 = surface water volume flowing out of the wetland at its downstream end

G2 = groundwater volume flowing out of the wetland

E = evapotranspiration volume

The analysis is based on annual averages. To determine the surface water outflows from the wetland it is necessary to perform the water budget analysis assuming a time interval of one year; this is the annual water budget. An assumption is also made that on an annual basis there is no change in the volume of water stored in the wetland ($\Delta S = 0$). Consequently, on an annual basis the inputs to a wetland are assumed to equal the outputs:

$$P + Q1 + QL + G1 + W = E + Q2 + G2$$

Manning's Equation Analysis

Manning's equation is commonly used to characterize flow conditions in open channels and in floodplains adjacent to the channel. The equation relates discharge (Q) to wetland slope (S), the roughness of the channel or wetland (n), the cross-sectional area of flow (A), and the length of the ground surface in contact with the water being discharged (i.e., wetted perimeter, P). Manning's equation is commonly written as:

$$Q = (1.49n^{-1})(A)(R^{2/3})(S^{1/2})$$

where:

R = hydraulic radius (A/P)

The equation is strictly applicable only under conditions of uniform flow in which the depth, water cross-sectional area, velocity, and discharge in a channel reach are constant. Uniform flow also requires that the energy gradient, water surface, and channel bottom have the same slope. In natural streams and particularly in wetlands, uniform flow rarely exists; however, the uniform flow conditions are often used in computations of flow characteristics in natural streams. Consequently, the use of Manning's equation must be viewed as a means of approximating flow conditions in wetlands, and is presented here as a simple mathematical tool for screening hydrologic changes in wetlands.

The Manning's equation analysis estimates the depth of flow in the wetland for a known discharge (Q), slope (S), roughness coefficient (n), and cross-sectional geometry. The slope and cross-sectiongeometry are determined from topographic maps and/or a site survey. The discharge (Q) is determined by the water budget analysis.

Data Collection and Compilation

The data required to support the water budget and Manning's equation analysis are generally obtained form published sources, from government data bases, and from a one-day wetland site survey. Table 1 summarizes the potential sources for information to support the analyses. The channel or wetland cross-section geometries generally include rectangular, trapezoidal, and triangular shapes that can be determined form topographic maps. The purpose of the one-day site reconnaissance is to produce a detailed map of the wetland topography. For a closed hydrologic system, vegetation should be studied to determine the approximate location of the annual average area-of-inundation. Values for the roughness coefficient are referenced in the handbook; however, photographs should be taken to assist in determining values for Manning's n.

APPLICATION OF ANALYSES

The three analyses can be applied to either open or closed wetland systems with a variety of different channel geometries. It should be noted that each wetland has a unique character. Channel shapes may vary, and the vegetation or physical parameters may change in some areas; therefore, it is important that the best judgement be used in identifying wetland characteristics.

The EPA handbook provides both a detailed

TABLE 1. DATA REQUIREMENTS AND SOURCES FOR A BASIC ANALYSIS

Component	Source
<u>Water Budget Analysis</u>	
Precipitation (P)	Local climatological data, annual and monthly summaries available from the NOAA National Climatic Data Center, Asheville, NC
Surface Water Inflow (Q_1)	U.S. Geological Survey Water Resources Data for the state of interest
Wastewater Applications Flowrate (W)	Specified in system design
Evapotransporation (E)	Local climatological data
Surface Water Outflow (Q_2)	Calculated as residual in the water budget analysis
Groundwater Flow (G_1, G_2)	Engineering judgment based on (1) County Soil Surveys published by the Soil Conservation Service; (2) geological and geohydrological reports by the U.S. Geological Survey and state Geological Survey
Wetland Area (A_w)	Topographic maps and site surveys
Drainage Areas	Topographic maps
Average Area-of-Inundation (closed hydrologic systems only)	Site surveys of vegetation distribution/type
<u>Manning's Equation Analysis</u>	
Manning's-n (n)	Site surveys, photographs
Wetland Slope (S)	Topographic maps or site surveys
Channel/Wetland Geometry	Site surveys

explanation of the methodology of the analyses and examples to illustrate the procedure. An example from the handbook of the results of a Basic Analysis for a hypothetical wetland is summarized in Table 2. Bill's Marsh, a 300-acre hydrologically open wetland, was evaluated for a proposed 1-mgd wastewater discharge. For three transects (A, B, and C), volumes of flow, depth and cross-sectional area, and velocity were calculated and averaged for existing conditions and for conditions with wastewater application. The differences between the two conditions for each

TABLE 2. SUMMARY OF HYDROLOGIC ANALYSIS RESULTS FOR BILL'S MARSH

Cross-Section	Flow (ft^3/sec)		Depth (ft)		Area (ft^2)		Velocity (ft/sec)	
	Exist	Appl	Exist	Appl	Exist	Appl	Exist	Appl
A-A'	75	77	0.84	0.84	971	971	0.077	0.077
B-B'	76	78	0.67	0.67	584	584	0.130	0.130
C-C'	77	79	0.54	0.54	1134	1134	0.068	0.068
Average	76	78	0.68	0.68	896	896	0.092	0.092

Change in depth = 0.00 inches. Minimal change.

Change in velocity = 0.00 ft/sec. Minimal change.

Area-of-inundation: Existing = 230 acres

Application = 230 acres

Change in area-of-inundation = 0%. Minimal change.

Residence time: Existing = 10.9 hours

Application = 10.9 hours

Change in residence time = 0%. Minimal change.

TABLE 3. DATA REQUIREMENTS FOR A REFINED ANALYSIS

Component	Method	Frequency
<u>Water Budget Analysis</u>		
Precipitation	Rain gauge	Weekly
Evapotranspiration	Class-A pan	Weekly
Groundwater Flow		
Permeability	Falling head permeability	Two times
Groundwater Level	Monitoring wells	Monthly
Surface Water Flow		
Water Level	Water level recorders	Continuously
Velocity	Current meter	Monthly
Area	Survey of channel	Monthly
Water Storage		
Water depth	Metal posts	Monthly
Wetland topography	Site survey	One time
<u>Manning's Equation Analysis</u>		
Manning's-n	Site survey	Winter and summer
Wetland slope	Site survey, topographic map	One time
Channel/wetland geometry	Site survey, cross-section diagrams	One time

of the parameters were evaluated to determine the impacts of 1 mgd of wastewater. The Basic Analysis reflects a minimal change in wetland hydrology. If there had been some significant changes to Bill's Marsh, then a Seasonal Analysis or a Refined Analysis would be conducted. Both analyses have procedures similar to the Basic Analysis, however, the data requirements and site survey are different. Table 3 illustrates the more extensive data requirements necessary for the Refined Analysis.

SUMMARY

The hydrology and hydraulic characteristics of wetlands must be considered in evaluating the use of wetlands for wastewater management. This paper summarizes the analyses developed

for the EPA's *Freshwater Wetlands for Wastewater Management Environmental Assessment Handbook*. The methods, based on certain hydrologic and hydraulic assumptions and generalized wetland characteristics, were developed to optimize the use of available local and regional information and minimize field work. The analyses utilize fundamental hydraulic principles to quickly and systematically evaluate the changes in wetland hydrology with wastewater application on a specified wetland.

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Wetlands: Toxicant Sinks or Reservoirs?

Mark L. Kraus
Hackensack Meadowlands Development Commission

INTRODUCTION

The concept that wetlands serve as nutrient and toxicant sinks is a long standing ecological "truth". Despite this generally accepted idea, many authors have begun to question its validity. Schubel and Kennedy (1984) refer to the estuarine filter as "leaky", and Valiela et al, (1976) suggest that estuaries may have upper thresholds associated with their toxicant assimilative capacity. Many studies have demonstrated that heavy metals accumulate in emergent estuarine vegetation (eg. Kraus et al, 1986; Sanders and Osman, 1985; Breteler et al, 1981; Banus et al, 1975). These studies suggest that the uptake and accumulation of heavy metals in leaf tissue provides a pathway out of the estuary. To compound this, other studies have shown that the saltmarsh cordgrass (Spartina alterniflora) is able to excrete a variety of nutrients (McGovern et al, 1979) including phosphorus (Reimold, 1972) as well as heavy metals (Kraus et al, 1986).

Ample data also exist to demonstrate that heavy metals are found in lower food chain animals from contaminated marshes (eg. Kraus, 1986; Callahan and Weis, 1983; Luoma, 1977; Ray and Tripp, 1976). Presumably the heavy metals were obtained through contact and feeding in and on "leaky" contaminated estuaries. This lower food chain export of heavy metals then has the potential to biomagnify.

The movement of heavy metals out of wetlands is not limited to estuaries. Wickland (1982) showed that cattails (Typha latifolia) in mine tailing ponds accumulated copper and zinc in their leaf tissue. This phenomenon has also been demonstrated for a variety of heavy metals in several aquatic plants (Mhatre and Chapekar, 1985; Franzin and McFarlane, 1980). In addition, as with their estuarine counterparts, fresh water invertebrates accumulate metals in their body tissues (Hildebrand et al, 1980; Fraser et al, 1978), which can also biomagnify up the food chain.

Despite the overwhelming evidence that both fresh and saltwater wetlands act as contaminant pumps, many authors still suggest that wetlands can be used for wastewater treatment. Breteler et al (1981) suggest that the application of metal containing "sewage sludge fertilizer" on estuarine wetlands does not increase mercury concentrations in cordgrass (Spartina alterniflora), or in a variety of invertebrates. In addition, Simmer et al (1987) states that the movement of heavy metals through vegetation into the surrounding environment will not be of concern when reclaiming contaminated dredge spoil materials with Spartina alterniflora. These authors do, however, suggest that uptake by animals may present a problem.

Similar to estuaries, freshwater wetlands have also been suggested for uses such as tertiary water treatment (Kadlec, 1976), and as detention basins for urban, highway, and industrial stormwater (Meyer, 1985). In addition, New York City is discussing plans to channel urban stormwater runoff into existing freshwater wetlands on Staten Island for treatment, in lieu of constructing stormwater systems (pers. comm. T. Hand, NYC Planning). Similar plans are being developed in Bellevue, WA (Bissonnette, 1987).

The purpose of this study was to determine what role emergent plants play in the uptake, and thus potential export, of heavy metals out of contaminated estuarine marshes within the Hackensack River Basin of Northeastern New Jersey. Four sites were chosen. Two of these sites were highly impacted areas in the towns of Carlstandt and Ridgefield. The third site was a newly initiated Spartina alterniflora mitigation site in the town of Secaucus, and the last site was a less severely impacted area of the Sawmill Creek Wildlife Management Area in Lyndhurst.

MATERIALS AND METHODS

Soil and plant samples were collected from each area. Although not all of the plant species were found at each site, above and below ground plant tissue was collected from the following species: Cordgrass (Spartina alterniflora), bulrush (Scirpus validus), cattail (Typha latifolia), and reed grass (Phragmites australis).

All samples were collected on June 1, 1987. Vegetation samples were thoroughly cleaned with tap water, then rinsed in distilled water. The plant material was separated by site and tissue type and allowed to air dry on brown paper sheets. After air drying, the samples were dried at 90°C overnight, then ground to a fine powder in a Waring blender.

Soil samples were also air dried on brown paper sheets. After air drying, the samples were dried at 90°C overnight. Oven dried samples were pulverized with a mortar and pestle,then passed through a 1 mm sieve.

All samples were prepared for heavy metal analysis by digesting dry, pulverized samples in analytical grade nitric acid (HNO_3) and perchloric acid ($HClO_4$). Samples were analyzed using atomic absorption spectrophotometry (Perkin-Elmer 272). Quality control was maintained by analyzing standards of known metal content and blanks, as well as National Bureau of Standards River Sediment (NBS 1645). Samples were analyzed for copper, nickel, cadmium, and lead.

RESULTS

The calculated recovery rate for the river sediment (NBS 1645) were 87.5%, 114.8%, 106.8%, and 96% for copper, nickel, cadmium, and lead respectively. These data indicate that the heavy metal contents reported in this study are not artifacts of the analytical techniques used.

Data are summarized in Table 1. In general, soil concentrations of the heavy metals analyzed were higher than the amounts found in the plant tissue. In most cases, (25/32; 78.2%) roots had the highest plant tissue accumulation of metal followed by rhizomes (5/32;15.6%), then stem or leaf tissue (1/32;3.1%).

DISCUSSION

The data reported in this study are not unique. They confirm what has been reported time and time again: that emergent vegetation in contaminated estuaries accumulates heavy metals. Although direct links between accumulation of heavy metals in above ground plant parts and their subsequent transport out of the estuary are generally lacking, Kraus et al (1986) demonstrated that S. alterniflora leaves retained mercury even after overwintering. It is not unreasonable to believe that the mercury, or other metals accumulated in overwintered leaves, would eventually enter the estuary associated with S. alterniflora detritus. Simpson et al (1983), working on freshwater tidal wetlands on the Delaware River, report similar findings. These authors found that the vegetation accumulated heavy metals throughout the growing season. In addition, dead, above ground plant material retained the metals. These authors felt that, due to the rapid decomposition of the plant material, the vegetation only served as temporary storage for heavy metals.

Although the metal levels in above ground plant material reported in this study are relatively low (mg/Kg or ppm), they represent levels well above what is considered background (see Moore and Ramamoorthy, 1984). Regardless of how low the vegetation export levels are, metals are moving out of the system. This suggests that wetlands may not really be sinks, but may actually be toxicant reservoirs.

This leads to some very interesting questions. For instance, how much metal enrichment above background is safe? Kraus et al (1986) showed that small amounts of mercury (0.02 mg/Kg) were present in S. alterniflora leaves from a relatively clean estuary which contained low amounts of this metal in the soil (0.22 mg/Kg $\pm$ 0.04). If even low concentrations of metals can move out of a wetland via the vegetation, should wastewater or urban runoff be channeled into wetlands for treatment? And what are the impacts of creating wetlands for stormwater retention/detention? It is obvious that wetland plants can absorb and accumulate metals in their above ground tissue. This is true for both woody and non-woody species (Wickland, 1982). If plants are capable of this, then using wetlands for wastewater purification may only serve to delay the process of releasing toxicants into the water column. A wetland could potentially absorb a pulse of pollutants, but it would slowly release the pollutants into the environment over time. The question then becomes whether simply delaying the pollution problem and possibly degrading the wetland is really the solution. Probably not, but it is the most cost effective solution in the short term.

Another potential problem associated with wastewater treatment in wetlands is the accumulation of metals in the roots and rhizomes of the wetland plants. Although probably not a significant problem with woody plants, the rhizomes of many non-woody emergent plants are important wildlife foods. Muskrats (Ondatra zibethicus) utilize both cattail (Typha sp.) (Weller, 1978) and reedgrass (Phragmites sp.) (Galluzzi, 1976) rhizomes as food. In addition, cattail rhizomes are an important food source for a variety of wild North American geese (Martin et al, 1951). The current study has shown that the below ground plant parts generally contain higher levels of heavy metals than do the above ground parts. This, coupled with the fact that rhizome consumers will also ingest contaminated soil in the feeding process, leads to the conclusion that these animals have the potential to fairly rapidly develop large body burdens of heavy metals. In addition to this, lower food chain animals readily accumulate heavy metals (e.g., Kraus, 1986; Callahan and Weis, 1983; Hildebrand, 1980). Many of these animals are important food sources for fish and birds. This becomes another major pathway for toxicants to move out of the system.

Although the intent of this paper is not to suggest that already contaminated wetlands should be completely filled and paved, it does point out that wetlands function better as toxicant

Table 1 - Distribution of copper, nickel, cadmium and lead in the soils and vegetation of the Mill Creek Mitigation Site, and in contaminated emergent estuarine wetlands in Carlstadt and Ridgefield, New Jersey. Data are also reported for a minimally contaminated site within the Sawmill Creek Wildlife Management Area. Data are reported in mg/Kg (dry weight). * denotes an average from a duplicate sample.

	COPPER	NICKEL	CADMIUM	LEAD
NBS 1645 River Sediment				
certified	109.0	45.8	10.2	714.0
recovered	95.4	52.6	10.9	685.3
Mill Creek				
*Soil	79.5	53.9	1.7	1091.5
Spartina alterniflora				
root	45.3	28.7	0.8	155.9
rhizome	44.9	3.5	0.5	25.0
leaf	14.9	0.5	0.8	6.5
Carlstadt				
*Soil	368.9	122.4	17.5	350.0
Phragmites australis				
root	116.8	46.4	16.5	104.8
*rhizome	7.2	5.7	0.8	10.0
*stem	7.2	3.5	1.0	11.2
leaf	7.8	0.0	1.0	5.0
Scirpus validus				
root	223.7	61.2	25.8	131.9
rhizome	247.5	33.9	3.5	20.0
leaf	23.4	11.5	1.0	20.0
Typha latifolia				
root	139.1	24.4	5.9	246.0
rhizome	92.4	33.0	11.5	414.4
*stem	10.3	8.8	1.8	30.0
leaf	8.5	6.0	1.0	19.9
Ridgefield				
soil	165.3	59.3	4.0	230.5
Spartina alterniflora				
root	21.4	0.0	1.6	0.0
rhizome	10.6	10.6	1.0	9.6
*stem	12.5	8.8	0.7	16.5
leaf	11.5	9.0	0.5	10.0
Phragmites australis				
root	18.5	12.5	8.0	25.0
rhizome	43.9	0.9	0.5	5.0
*stem	6.5	8.2	0.3	5.0
*leaf	8.7	10.5	0.8	12.2
Scirpus validus				
root	150.4	54.8	99.6	43.3
*rhizome	46.6	23.4	1.7	24.9
leaf	16.0	9.0	0.5	5.0
*flowers	19.1	9.4	0.5	12.4
Sawmill Creek				
Soil	16.0	11.0	0.0	30.0
Phragmites australis				
root	34.0	13.0	2.0	15.0
rhizome	10.0	5.0	0.0	5.0
stem	9.0	2.0	1.0	0.0
leaf	10.0	3.0	0.0	0.0

reservoirs, rather than toxicant sinks. One of the very important functions of wetlands is their ability to act as nutrient/toxicant retention systems. Wetlands perform this function very well under natural conditions, but the question remains as to whether wetlands can function in this capacity under abnormal enrichment. The data presented in this study indicate that sewage and/or urban runoff wastewater should not be discharged into wetlands for treatment. These data may have further implications with respect to the siting of mitigation/enhancement sites and dredge spoil reclamation projects. These data also raise questions as to the design of stormwater retention/detention basins.

In summary, we should be very careful not to sacrifice some wetland functions in order to capitalize on others. Yes, wetlands can absorb pulses of nutrients or toxicants under natural conditions, but under abnormal enrichment there is a strong possibility that the wetland may lose some of its assimilative capacity, and that other wetland functions may suffer.

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Hydrology and Water Quality of a Wetland Used to Receive Wastewater Effluent, St. Joseph, Minnesota

J.R. Stark and R.G. Brown
U.S. Geological Survey
St. Paul, Minnesota

INTRODUCTION

Small rural communities are using wetlands as cost-effective alternatives to tertiary treatment of wastewater. Chemical constituents in secondarily treated wastewater are assumed to be partially retained in the wetland through physical and biological processes.

In 1980, the U.S. Environmental Protection Agency identified 28 municipalities in Minnesota that discharge secondarily treated wastewater to wetlands for tertiary treatment (WAPORA,1983). The St. Joseph wetland, in central Minnesota, (fig. 1) was selected to evaluate the effects of wastewater discharge on a northern wetland. The town of St. Joseph has discharged 900 m^3/d (cubic meters per day) of secondarily treated wastewater into the wetland during 1962-86. The municipal sewage-treatment plant, located north of the city (fig. 1), serves a population of 3,000 permanent residents and 1,700 college students.

PURPOSE AND SCOPE

This paper discusses the hydrologic and chemical balances of the wetland and the vegetational changes since wastewater discharges began, as part of the research at the St. Joseph wetland. The findings are preliminary, and additional data collection and analysis and model simulation are planned for the next several years of research.

PHYSICAL AND HYDROLOGIC SETTING

The wetland is located northwest of the St. Joseph wastewater-treatment plant and covers about 0.186 km^2 (square kilometers) (fig. 2). About half of the wetland area consists of tamarack bog adjacent to a cattail marsh (fig. 2). Grassland and isolated stands of hardwood forest surround the wetland. The surface-water drainage area of the wetland is 0.98 km^2, of which 0.94 km^2 is drained by a storm sewer that collects runoff from the urbanized part of St. Joseph. Discharge from the sewage-treatment plant, overland runoff, and discharge through the storm sewer are the major sources of surface water to the wetland. The wetland discharges through a small channel to the South Fork of the Watab River approximately 400m to the northwest.

The wetland is located on a glacial-outwash plain. Surficial geologic materials in the area consist of outwash, scattered areas of clay-rich till surrounded by outwash, alluvium associated with river valleys, and wetland deposits (fig. 1). The outwash is from 0 to 20 m thick and is thickest in the immediate vicinity of the wetland. The outwash consists primarily of medium to coarse, well-sorted sand; it is underlain by clay-rich till that is as much as 30 m thick. The till is, in turn, underlain by silty sandstone that overlies granite bedrock of the Precambrian (Kanivetsky, 1978).

HYDROLOGIC BALANCE

Data were collected from October 1985 through September 1986 to determine the monthly hydrologic and chemical balances of the bog and to improve quantification of ground-water and surface-water interactions within the bog and marsh.

Monthly hydrologic balances for the bog and marsh were calculated using the following equations:

Bog equation: $(P + SSI + WWI + GWI) - (ET + SWO) = \pm S$ (1)

Marsh equation: $(P + SWI + GWI) - (ET + SWO) = \pm S$ (2)

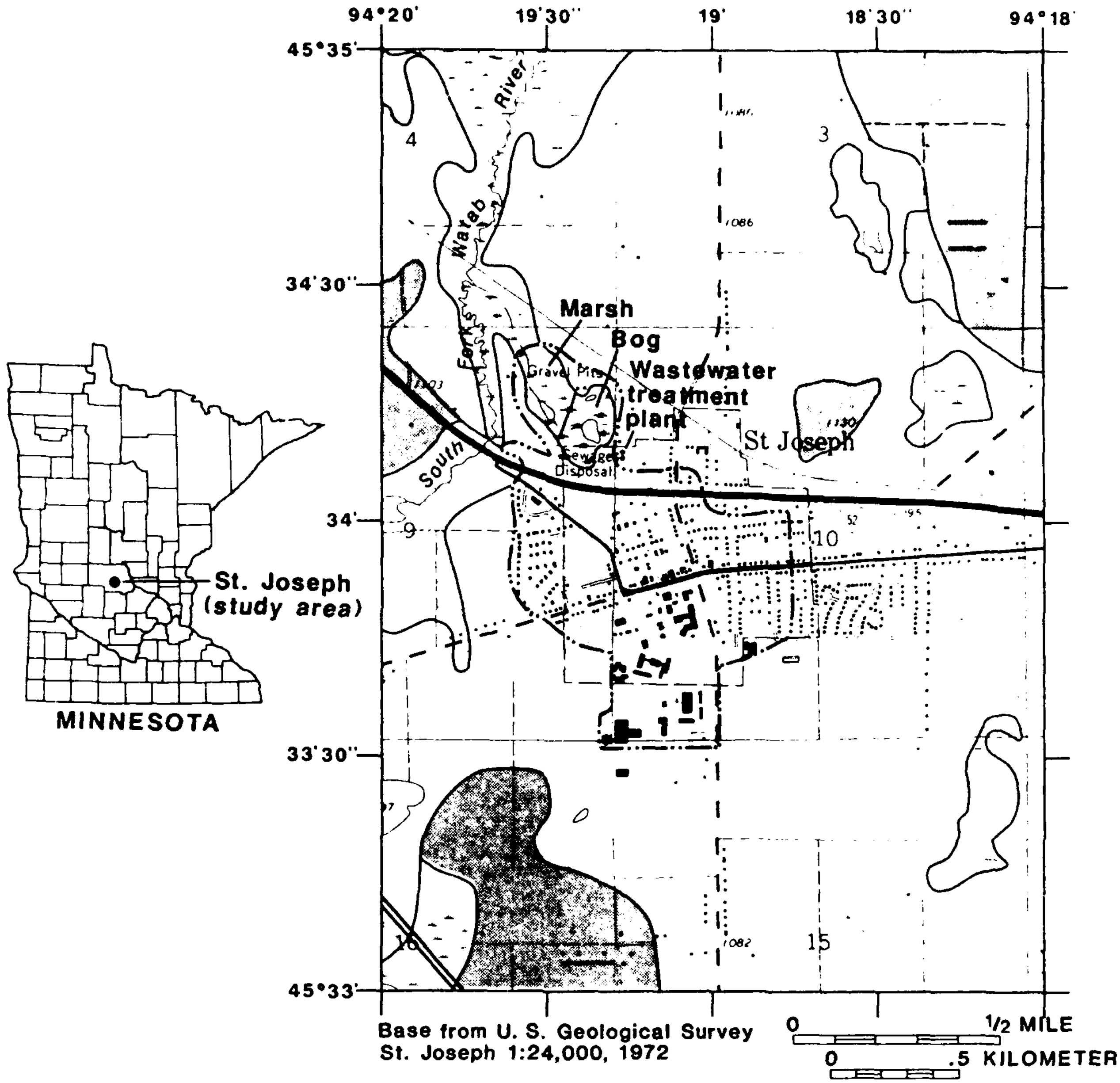
where inflow components are:

- P - incident precipitation on the bog or marsh in cubic meters per month;
- SSI - storm-sewer inflow to the bog, in cubic meters per month;
- WWI - wastewater inflow to the bog, in cubic meters per month;
- GWI - ground-water inflow to the bog, or marsh in cubic meters per month;
- SWI - surface-water inflow to the marsh, in cubic meters per month (which is the same as the surface-water outflow from the bog);

and where outflow components are:

- ET - evapotranspiration from the bog of marsh, in cubic meters per month;
- SWO - surface-water outflow from the bog or marsh, in cubic meters per month;

and where the storage component is:



EXPLANATION

WATERSHED BOUNDARY

AREA OF WATERSHED DRAINED BY STORM SEWERS

SURFICIAL GEOLOGY:

GLACIAL OUTWASH

SWAMP AND WETLAND DEPOSITS

GLACIAL TILL

ALLUVIUM

Figure 1.--Location and physical features of study area

S = change in surface-water in the bog or marsh, in cubic meters per month.

All components of the hydrologic balance were measured or estimated from measurements made in the field (fig. 2). Ground-water inflow was estimated using a ground-water flow model developed by McDonald and Harbaugh (1984) and compared to inflow estimates calculated as a residual from the hydrologic-balance calculations. The wetland is a ground-water discharge area with flow from all directions. Horizontal ground-water gradients average about 0.002. In the area of the wetland, the ground-water system has a relatively large vertical upward gradient of 0.1. Ground water is confined or partially confined beneath the wetland by organic materials that have a vertical hydraulic conductivity about 2 percent of that of the horizontal hydraulic conductivity of the outwash. These findings enabled hydrologic-balance and chemical-balance calculations to be made by treating ground-water inflow to the wetland as a residual, because ground-water outflow from the wetland was shown to be negligible. Simulations included values of recharge to the wetland of 0.60 m per year, evapotranspiration of 0.15 m per year, and vertical hydraulic conductivity of the surficial aquifer of 30.5 m/d (meters per day) (Helgesen, Ericson, and Lindholm, 1975). These assumed values resulted in simulated potentiometric surfaces and vertical hydraulic gradients that approximated field measurements. Results from these simulations were used to estimate ground-water inflow to the wetland by summing flow rates to streambed cells representing the wetland. Although these calculated values are highly sensitive to modeled values of recharge, vertical hydraulic conductivity of the organic deposits, and hydraulic conductivity of the aquifer, they were used as a means to evaluate ground-water values calculated as a residual from hydrologic-balance estimates. Additional refinement of the model is planned for future work.

Surface-water inflow from the part of the surface-water drainage basin that is not storm sewered (fig. 1) is assumed to be insignificant because (1) the drainage area is small (0.04 km^2) and (2) surface runoff is minimal because of permeable soils and low relief.

Precipitation was measured with a tipping-bucket rain and snow gage located 60 m southeast of the bog (fig. 2).

Evapotranspiration was estimated from daily solar radiation, daily maximum and minimum air temperatures, and daily average air humidity data using methods described by Jensen and Haise (1963) and Jensen and others (1969). In this study, evapotranspiration is equal to potential evapotranspiration because of continuously saturated conditions.

Wastewater discharge was calculated using a Parshall flume and stage data (fig. 2). Storm-sewer inflow, bog outflow (which is also the marsh inflow) and marsh outflow were calculated from stream-stage measurements using a relationship between stage and measured discharge (Kennedy, 1984).

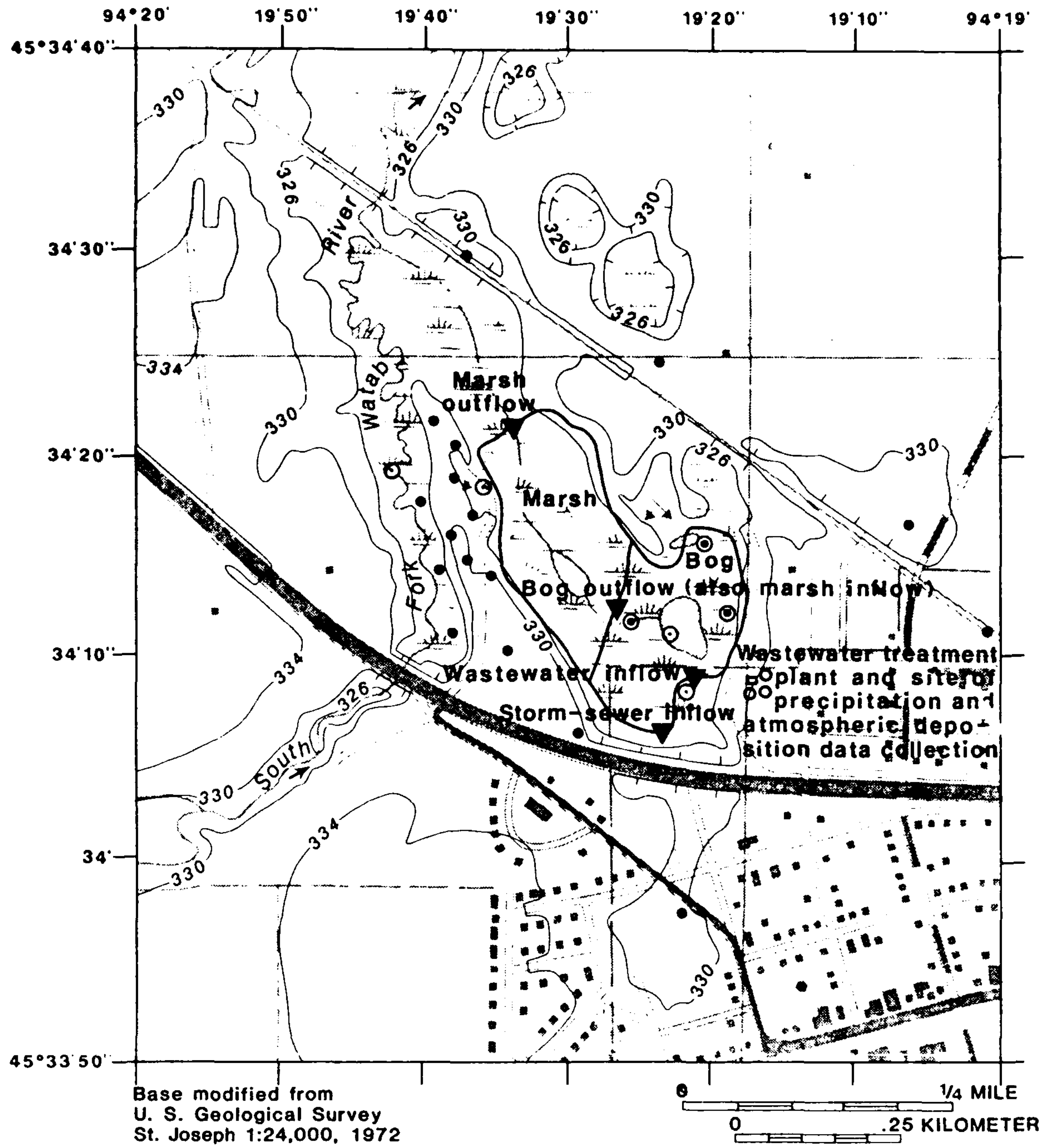
Changes in storage within the bog and marsh were measured using data from a topographic map and water-level data collected at the bog and marsh outflows. Errors associated with calculation of the hydrologic-balance components were estimated using methods described by Winter (1981). The errors, calculated as the standard error of measurement, are expressed as the average percent error in measurement of the component as follows: (1) 8-percent error in precipitation; (2) 28-percent error in evapotranspiration; (3) 9-percent error in surface-water inflow from the storm sewer and wastewater; (4) 14-percent error in surface-water outflow (including marsh inflow); (5) 41-percent error (error of the residual) in the groundwater inflow; and (6) 11-percent error in change in storage. These errors in measurements for each component represent qualitative estimates.

DISTRIBUTION OF HYDROLOGIC COMPONENTS

The dominant inflows to the bog generally were from ground-water and wastewater. Wastewater inflow is larger during the winter months because of an increase in population in the town. Surface-water inflow (predominantly storm-sewer discharge) is significant only during summer months. Storm-sewer discharge during April was primarily from snowmelt. Total inflow to the marsh was predominantly surface-water outflow from the bog. The distribution of annual hydrologic-balance components for the bog and marsh is given in Table 1.

The bog and marsh are both ground-water-slope wetlands, defined as wetlands located in the headwaters of surface-water-drainage basins which receive a significant part of inflow from the ground-water system and in which surface water is the dominant outflow (Novitzki, 1978). Discharge of wastewater into the bog affects the hydrology of both the bog and the marsh. If wastewater was not discharged into the bog, ground-water inflow would be a greater percent of total inflow to the bog and marsh.

Independent estimates of ground-water inflow to the wetland were made using model results and the residual from hydrologic-balance calculations. Results indicate that model-calculated discharge rates are about half of the ground-water inflow rate calculated by the budget residual method. Although model-calculated values are highly sensitive to values of recharge, vertical hydraulic conductivity of the organic deposits, and hydraulic conductivity of the aquifer, they do suggest that ground water



EXPLANATION

—330— TOPOGRAPHIC CONTOUR LINE
Interval 4 meters. Datum is mean sea level

▼ SURFACE-WATER DATA-COLLECTION SITE

WELL SITES:

● OBSERVATION WELL

⊙ NEST OF OBSERVATION WELLS

◉ NEST OF OBSERVATION WELLS IN WHICH A WELL WAS SAMPLED FOR WATER-QUALITY ANALYSIS

Figure 2.--Location of wetland areas and data-collection sites

Table 1. Distribution of components in hydrologic balance for the bog and marsh during 1986 (--, not applicable values are percentages of total annual budget, inflow or outflow)

Hydrologic-balance component	Bog	Marsh
Inflow		
Wastewater	38	--
Surface water	12	74
Precipitation	8	4
Groundwater	42	22
Outflow		
Surface water	92	92
Evapotranspiration	6	6
Storage	2	2

values calculated as a residual probably are in reasonable agreement with values calculated with the model.

CHEMICAL BALANCE

The monthly chemical balance for the bog and marsh were calculated using the following equations:

Bog equation: $(A + SSI + WWI + GWI) - (SWO) = \pm S$
Marsh equation: $(A + SWI + GWI) - (SWO) = \pm S$

where input components are:

A - atmospheric deposition to the bog, or marsh in kilograms (kg) per month;
SSI - storm-sewer input to the bog, in kilograms per month;
WWI - wastewater input to the bog, in kilograms per month;
GWI - ground-water input to the bog, or marsh in kilograms per month;
SWI - surface-water input to the marsh, in kilograms per month (which is the same as the surface-water output from the bog);

where output components are:

SWO - surface water output from the bog or marsh, in kilograms per month; and

and where storage components are:

S - change in storage within the bog or marsh, in kilograms per month.

Monthly chemical balances in the bog or marsh were calculated for total suspended solids, total phosphorus, and total ammonia plus organic nitrogen by multiplying the flow volumes by the concentration in water samples collected for each component, except storage, which was calculated as the budget residual. Samples were analyzed using methods described by Fishman and Friedman (1985).

Atmospheric deposition was estimated by multiplying monthly total precipitation by the constituent concentration in monthly samples of atmospheric deposition (wetfall and dryfall). The atmospheric-deposition samples consisted of material collected during the month in an atmospheric sampler mounted near the precipitation gauge (fig. 2).

Wastewater inflow, bog outflow (also marsh inflow), and marsh outflow were sampled biweekly (fig. 2) for analysis of each constituent. Storm-sewer inflow was sampled during five storms using flow-weighted composite samples of each storm with the method described by Nelson and Brown (1983). The monthly input or output was calculated from concentration and discharge data associated with each biweekly or storm sample using methods described by Nelson and Brown (1983).

Ground-water input was estimated by multiplying the monthly total ground-water inflow by the monthly average concentration of the constituent determined in bimonthly samples taken from three observation wells located in observation-well nests (fig. 2). The sampled wells were screened in the outwash aquifer underlying the peat in the bog and marsh.

The storage value was the difference between the input and output of a constituent during the month. A positive storage value indicated that there was a net retention of the constituent in the wetland during that month, while a negative value indicates a net release (output greater than input).

Errors associated with calculation of each chemical-balance component were estimated using methods described by Winter (1981), and represent the error of measurement, in percent, of the component as follows: (1) 32-percent error in atmospheric deposition; (2) 14-percent error in surface-water inputs from the storm sewer and wastewater; (3) 22-percent error in surface-water outputs (including marsh input); (4) 58-percent error in the ground-water input; and (5) 69-percent error (error of the residual) in change of storage. These errors in measurement for each chemical-balance component represent qualitative estimates.

DISTRIBUTION OF CHEMICAL COMPONENTS

Wastewater and storm-sewer inputs are generally the dominant chemical constituent inputs to the bog. Wastewater inputs were generally greater during October through April because of the presence of the student population during those months. Storm-sewer and atmospheric-deposition inputs occurred primarily during April through September. The monthly output of constituents (entirely associated with surface water) was consistently lower than the

monthly input; the difference was stored or retained. Surface water dominated chemical input to the marsh. Similarly, output of chemical constituents from the the marsh was consistently lower than input during the 12-month period. Storage or retention, of constituents was greatest during April through September. The distribution of components in the chemical balance for the bog and marsh during the year are shown below. Data on nitrogen are preliminary. Nitrogen components do not include nitrate and nitrite. A mass balance on nitrogen has not been completed.

Particulate or undissolved phosphorus is removed from the water by the same processes discussed for suspended solids (Boto and Patrick, 1978). These mechanisms have been postulated in previous studies as: (1) precipitation or sorption of phosphorus on organic matter (Spangler and others, 1977) and (2) assimilation of phosphorus by flora and fauna (Kitchens and others, 1975). Removal of phosphorus from the water is highly dependent on the wetland hydrologic regime. A low velocity of flow through a wetland is essential for net accumulation of particulate phosphorus in the litter and assimilation of dissolved phosphorus by plants (van der Valk and others, 1978). Removal or retention of dissolved phosphorus depends on a long residence time of water in the wetland, which is required for biological processes to occur (Klopatek, 1975).

Table 2. Distribution of components in chemical balance for the bog and marsh during 1986 (SS, total suspended solids; P, total phosphorus; NH_4, total ammonia nitrogen; N, total organic nitrogen; --, not applicable; values are *percentages of total annual input or output*)

Chemical-balance component	Bog			Marsh		
	SS	P	NH_4+N	SS	P	NH_4+N
Inputs						
Wastewater	73	96	81	--	--	--
Surface water	22	2	16	95	97	84
Atmospheric	4	2	2	4	2	14
Ground water	<1	<1	<1	<1	<1	<1
Outputs						
Surface water	66	86	86	56	82	78
Storage	34	14	14	44	18	22

RETENTION OF CHEMICAL COMPONENTS

Retention of suspended solids by deposition in a wetland is directly related to its flow characteristics. The velocity decrease, along with the presence of vegetation, promotes deposition of suspended solids (Boto and Patrick, 1978). Average monthly or annual retention of suspended solids is 34 percent of the input in the bog and 44 percent of the input in the marsh. The residence time of water in the bog and marsh is approximately 1.8 and 3.1 days, respectively. The more dense vegetational composition and longer retention time in the marsh promotes settling of suspended solids at a greater rate than in the less densely vegetated and partly open-water bog. The bog retains suspended solids primarily by decreasing velocity through ponding, while retention in the marsh is by a decrease in velocity through ponding and by the filtering action of the cattails. Greater retention during the months of April through September (spring and summer) is because these are active months for vegetation.

Retention of total phosphorus or removal of the constituent from the water as it passes through the bog and marsh is probably through sedimentation, as the residence time of water in the bog and marsh (1.8 and 3.1 days, respectively) generally is too short for removal of dissolved phosphorus. The part of total phosphorus that is retained is most likely associated with suspended solids that are retained in the bog and marsh. Annual retention of total phosphorus is 14 percent of the input in the bog and 18 percent of the input in the marsh. Greater retention during April through September (from monthly data not published in this report) is related to vegetational activity similar to that for suspended solids.

Particulate nitrogen is removed from the water in a wetland through the same processes discussed for suspended solids (Boto and Patrick, 1978); the major mechanism being dentrification (Klopatek, 1975; Lee et al, 1975). Retention of total

ammonia plus organic nitrogen, or removal of the constituent from the water as it passes through the bog and marsh, is probably similar to that for total phosphorus. Annual retention of total ammonia plus organic nitrogen is 14 percent of the input in the bog and 22 percent of the input in the marsh.

CHANGES IN VEGETATIONAL COMPOSITION

Vegetation in the bog and marsh appears to have been affected by the wastewater discharge. The vegetational composition in 1958 was 30-percent spruce, 30-percent tamarack, and 40-percent wetland grass in the bog, and about 30-percent tamarack and 70 percent wetland grass in the marsh (Williams, 1985). Vegetational composition in 1986 was 5-percent spruce, 45-percent tamarack, 15-percent sedge grass, 20-percent duckweed, and 15-percent cattail in the marsh. The change in vegetational composition between 1958 and 1986 was likely the result of changes in both the hydrologic and chemical balances of the bog and marsh as a result of the wastewater discharge. Introduction of cattails into both the bog and marsh is an indication of how the ecosystem has been altered between 1958 and 1986. Cattails have been shown to thrive in situations where wastewater is introduced. The eventual outcome is a monoculture of cattails in wetlands that receive wastewater (Godfrey et al, 1985).

SUMMARY AND CONCLUSIONS

The hydrologic and chemical balances of the bog and marsh in the St. Joseph wetland are greatly affected by the discharge of wastewater. Wastewater inflow represents 38 percent of the total inflow to the bog. If wastewater was not discharged to the bog, ground water would be a more dominant inflow component to both the bog and marsh, and the total inflow and outflow of both the bog and marsh would be approximately 38 percent smaller. The wastewater input of total suspended solids, total phosphorus, and total ammonia plus organic nitrogen represented 74, 96, and 82 percent of the total chemical-constituent input of total suspended solids and total phosphorus that exceeded the input from sources other than the wastewater. Therefore, if wastewater was not present, the export of these two constituents to the marsh would be nearly zero. The retention of total ammonia plus organic nitrogen would be primarily in the bog if wastewater input was not present.

The vegetational composition of the bog and marsh has changed greatly since discharge of wastewater began in 1962. The vegetational composition in the bog was a spruce/tamarack/wetland/grass bog in 1958 and a tamarack/cattail/wetland/grass bog in 1986. The marsh was a wetland-grass/tamarack marsh in 1958 and is a cattail/tamarack marsh in 1986. The invasion of cattails and loss of original vegetation may be an indication of the effects of wastewater input to the bog and marsh.

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Differences in Oxygen Productivity Rates and Reaeration Coefficients For Wetland Reaches of Natty Pond Brook, Massachusetts

Gene W. Parker and Nancy C. Suurballe, Hydrologists
U.S. Geological Survey

INTRODUCTION

Background

Stream-water quality has received a great deal of national attention over the past 25 years. Assessment of the wasteload-assimilation capacity of stream systems is required by various Federal, State and local agencies to maintain or restore stream water-quality standards. Assessment of the waste-assimilation capacity of a wetland-bounded stream is more difficult than for a normal riverine system. The effects of the wetland's natural conditions on stream-water quality must be understood before an attempt is made to simulate the effects of additional waste loads. Increased understanding of the effects of wetland processes on stream water-quality can help environmental management agencies protect and preserve natural resources. This report describes the results of a U.S. Geological Survey study to determine net daytime oxygen-productivity rates (referred to as oxygen-productivity rates in this report) and reaeration coefficients of three palustrine wetland reaches of Natty Pond Brook in Hubbardston, Massachusetts (fig. 1) and how they relate to differences in wetland types, water-quality conditions, and channel characteristics. This report also describes the use of measured reaeration coefficients to improve the accuracy of oxygen-productivity rates. The study was conducted in cooperation with the Massachusetts Department of Environmental Quality Engineering, Division of Water Pollution Control.

Reaeration coefficients generally are estimated from any of several equations that are used in the Stevens and Jennings model (1976) for the determination of oxygen-productivity rates. In a recent study, the error in estimation of reaeration coefficients for low-slope rivers by using 19 predicting equations ranged from 53 to 4,200 percent (Parker and Gay, 1987).

By use of the steady-state, gas-tracer method (Yotsukura and others, 1983 and 1984; Kilpatrick and others, 1987), reaeration coefficients were measured for each study reach during August and September 1985 and June 1986. These field studies were conducted during similar steady, approximately average annual flow conditions with respect to nearby gaging stations. The use of measured reaeration coefficients significantly improved the accuracy of determinations of oxygen-productivity rate as well as the confidence in the accuracy of subsequent analyses.

Physical Setting and Study Reaches

The Natty Pond Brook wetlands cover two of the 14 squarekilometers of the Natty Pond Brook basin. The vegetation community of the study basin consists 60 percent forested species, 30 percent scrub-shrub species, and 10 percent emergent and mixed vegetation or open-water plant types (fig. 2). Three reaches of Natty Pond Brook were chosen to represent different combinations of physical and hydrologic characteristics.

The water-quality in each study reach is directly related to the ratio of wetland area to the total drainage area. Only 2 percent of the total drainage of the upper reach is wetland. The middle reach has a 7 percent ratio of wetland to total drainage. Thirteen percent of the new drainage below the upper reach is wetland area. The lower reach has 12 percent wetland drainage within the total drainage. The wetland area is 23 percent of the new drainage below the middle reach and represents the largest ratio of wetland to total drainage for the study area.

The vegetative and physical characteristics of the individual reaches influence the ability of sunlight to reach the water surface and the energy level within the water column. The upper reach is through a scrub-shrub wetland and has a water-surface slope of 0.00004 meters per meter. The middle reach is through a predominately forested wetland area with a water-surface slope of 0.005 meters per meter. The lower reach is similar in local characteristics to the upper reach in that it flows through a predominately scrub-shrub wetland area and has a water-surface slope of 0.00004 meters per meter.

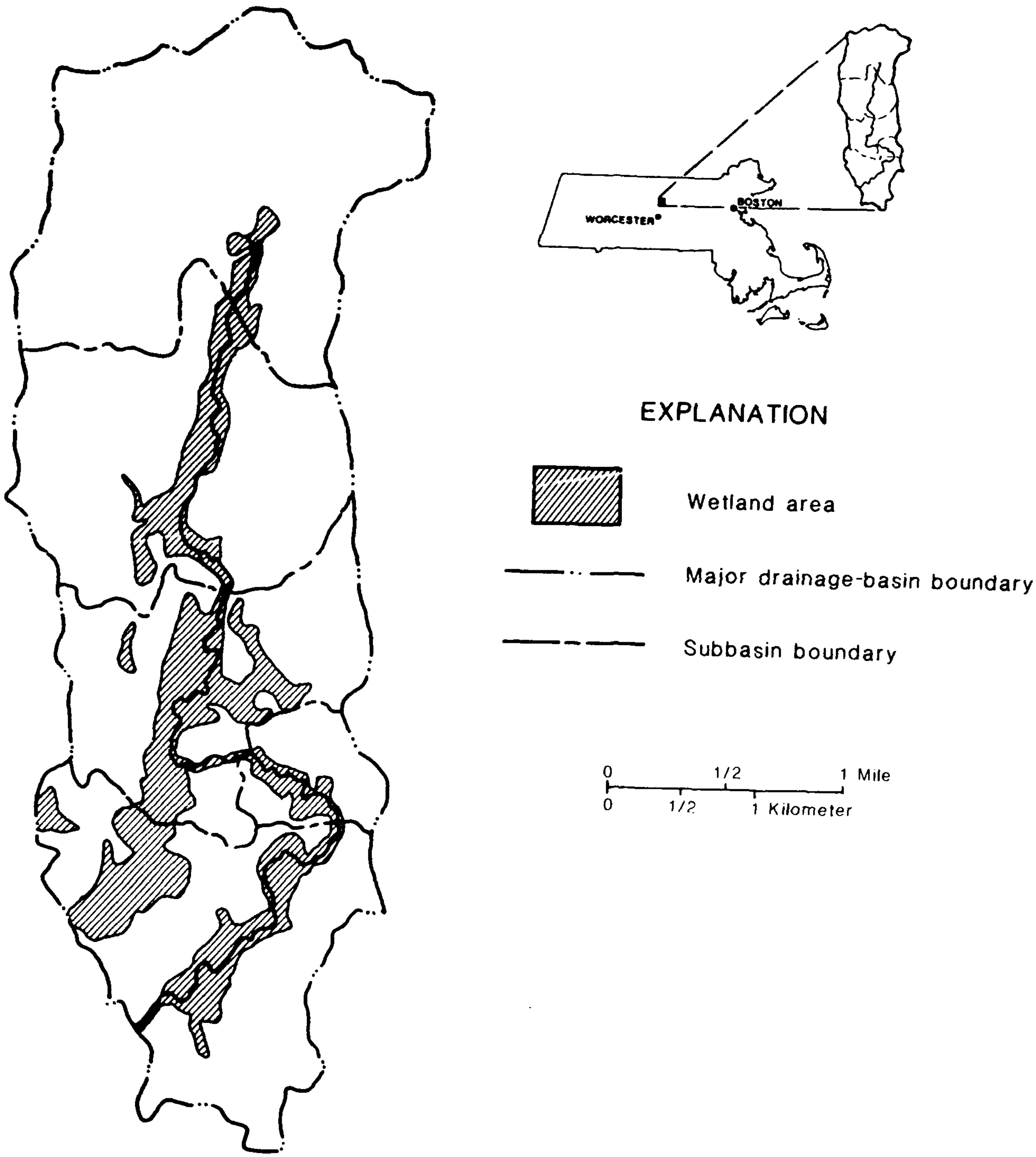

Figure 1.--Natty Pond Brook Basin, Hubbardston, Massachusetts

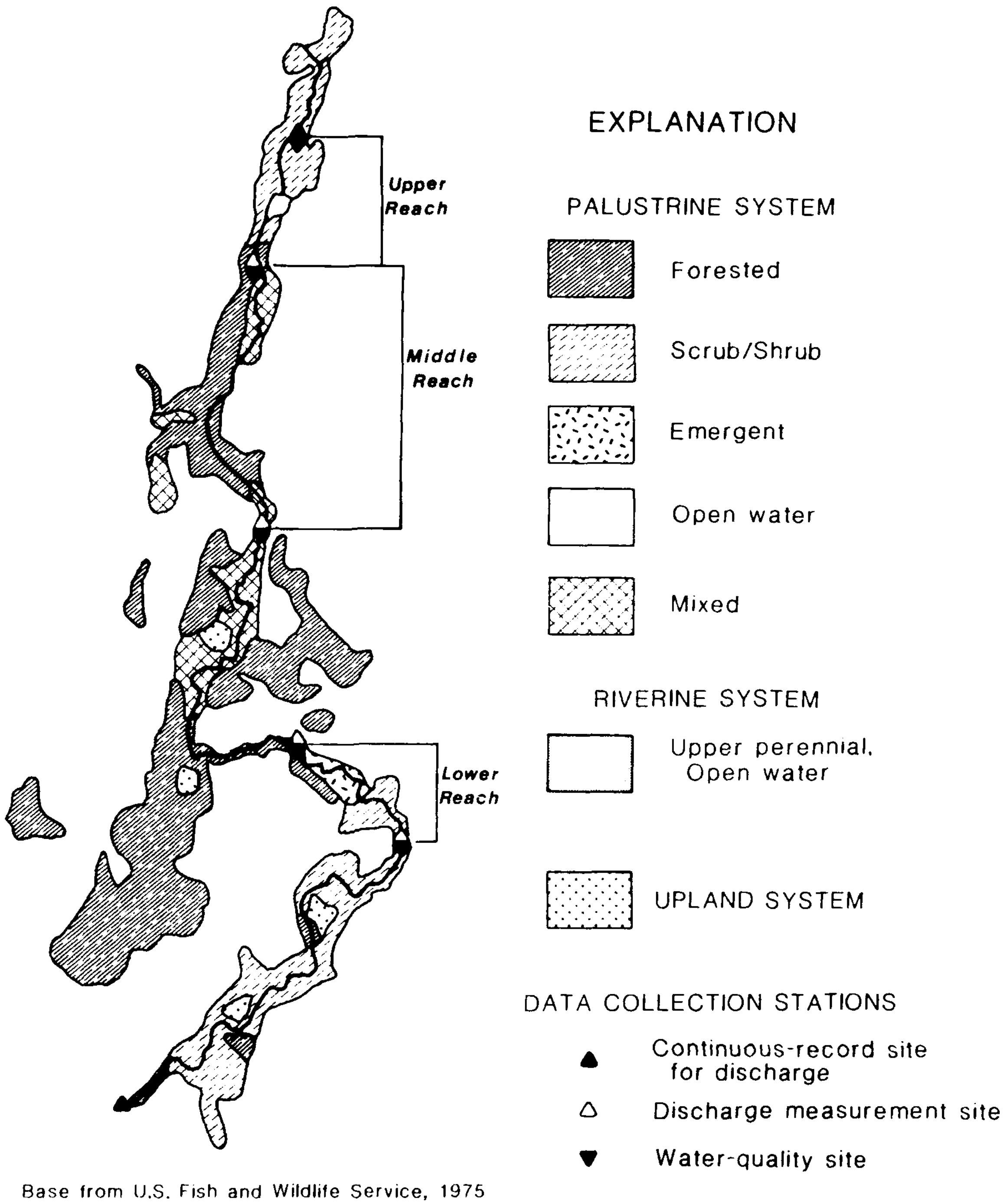

Figure 2.--Wetland types and study reaches on Natty Pond Brook, Hubbardston, Massachusetts

THEORY AND APPROACH

Determination of Net Daytime Oxygen-Productivity Rates

The oxygen-productivity rate (P) can be determined from a diel series of dissolved-oxygen and water-temperature measurements using an approach developed by Odum (1956) that was coded into a computer program by Stephens and Jennings (1976). For a single station, the program solves the oxygen-balance equation:

$$X = P - R \pm D \pm I \qquad (1)$$

where X is the rate of dissolved-oxygen concentration change, R is the rate of oxygen respiration, D is the oxygen-diffusion rate, and I is the rate of oxygen available from or the rate of oxygen demand by drainage accrual. The Stephens and Jennings (1976) program assumes that dissolved-oxygen use to or from drainage accrual is negligible if compared with the other components in eq. 1. If the drainage accrual is not negligible, the effects can be identified by comparing the results from the Stephens and Jennings program for each reach. During the night, any change in dissolved-oxygen concentration after correcting for oxygen diffusion through the water surface-film is the result of respiration, which is constant for each 24-hour period. The program also assumes that oxygen-production from periphyton and phytoplankton only occurs during the day and that any change indissolved oxygen, after correcting for diffusion, is the result of this daylight production minus respiration.

The determination of the oxygen-productivity rate depends on accurate measurement of the changes in dissolved-oxygen rate at a site and the accurate determination of a reach's oxygen-diffusion constant. The diffusion rate is a constant at zero-percent oxygen saturation and is calculated from:

$$D = k_2(9.07)/(BP/760) \qquad (2)$$

where k_2 is the reaeration coefficient for the reach, in units per day; 9.07 is the dissolved oxygen saturation at 20 degrees Celsius, in milligrams per liter; and BP is the barometric pressure, in millimeters of mercury.

Steady-State, Gas-Tracer Method

Theory

Reaeration coefficients were determined using the steady-state gas-tracer method described by Yotsukura and others (1983; 1984). The steady-state method involves concurrent injection of gas- and dye-tracers into the stream to be measured. Basically, this method measures the desorption of the propane against the more conservative dye-tracer. This method directly measures the depth-averaged desorption coefficient of propane gas, which then can be related to the reaeration coefficient. Propane is not naturally found in the environment, thus its use as a gas tracer eliminates any local interferences and isolates the study to measurement of only the transfer process. The gas- and dye-tracer injections were made at the same location upstream from the reach to ensure complete transverse mixing of the tracers by the time they entered the reach. Following complete transverse mixing, desorption is measured between the sections which comprise the test reach. All measurements were made under conditions of little- or no-wind to minimize the effects of wind shear on the desorption rate of gas through the water-surface film.

Dye-tracer studies

The slug-injection, dye-tracer method (Hubbard and others, 1982) was used to determine the time of travel and dispersion of the tracer through each reach. Steady streamflow conditions are required for this method. A measured volume of a 20-percent solution of rhodamine-WT -- a fluorescent dye -- was slug injected upstream from the reach. Water samples were collected at sites at the upstream and downstream ends of the reach to determine changes in dye concentration over time. Water-sample collection continued at each site until field analyses indicated that the dye concentration had dropped to 2 percent of the maximum concentration observed at that site. All water samples were maintained and reanalyzed at a constant temperature in the US Geological Survey's Massachusetts Office. Graphs of dye concentrationover time since injection were used to define the dye-response curve for each sample site.

At each sample site, the time of travel since injection was determined for four dye-response curve characteristics, as outlined by Parker and Hunt (1983). The characteristics are (fig. 3):

(1) Leading edge - The first detectable dye concentration observed at a sample site;

(2) Peak - The maximum dye concentration observed at a sample site;

(3) Centroid - The center of mass of the dye response curve observed at a sample site; and

(4) Trailing edge - The point on the falling limb of the dye response curve arbitrarily selected to equal 2 percent of the peak concentration observed at a sample site.

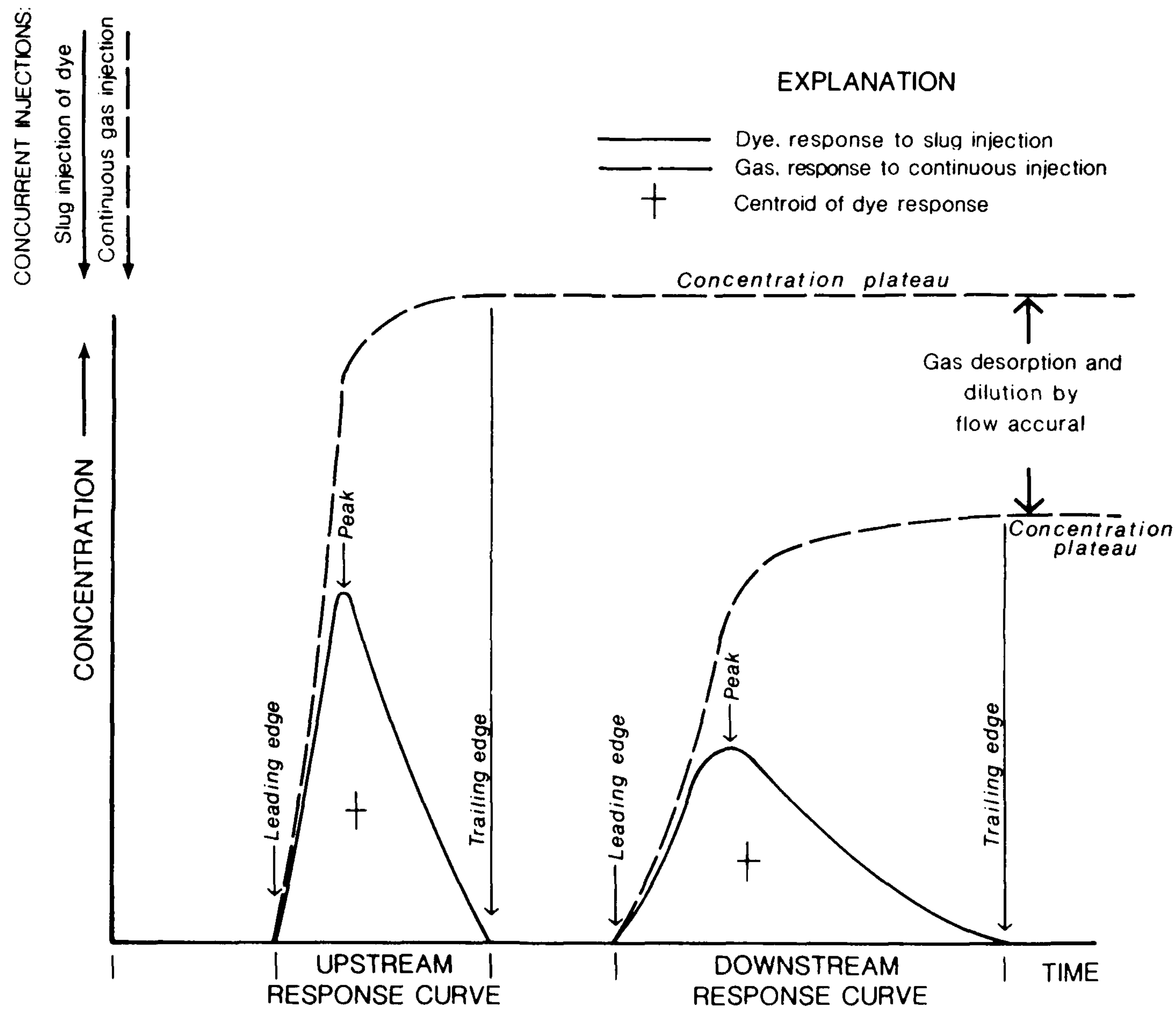

Figure 3.--Definition and characteristics of dye- and gas-tracer responses (modified from Kilpatrick and others, 1987, fig.5, p.16)

Gas-tracer tests

The gas-tracer test involves the steady, long-term injection of a commercial grade, propane-gas tracer (in contrast to the slug injection of the conservative dye-tracer) to determine the gas-tracer mass desorbed from the stream along a reach. The gas tracer is injected through a flat-plate, porous-tile gas diffuser placed on the river bottom. The number of diffusers used in a study was three or four, depending on the cross-section dimensions and flow conditions. Commercial-grade propane was injected from a 100-pound tank through a single-stage regulator and a rotameter. In all studies, gas-tracer injection was continuous for 24 to 72 hours depending on the results of the dye-tracer tests. Gas injection was maintained until all samples for propane analysis were collected.

Samples were collected to determine the concentration of the steady-state gas-tracer plateau at two sites. At any point along a river reach, the start of the concentration plateau for a continuously injected gas-tracer occurs at the same time as the arrival of a response curve's trailing edge for a concurrent slug-injected dye-tracer (fig. 3). For this reason, gas samples at each sampling site were collected only after the dye cloud's trailing edge had passed. The time differential between dye peaks was used to determine the interval between sites for gas sample collection. The propane-gas concentration in each sample was determined by gas-chromatography.

Calculations of reaeration coefficients

The gas-desorption coefficient (Kp) was calculated using the superposition principle outlined by Yotsukura and others (1983), which uses the gas-plateau concentration and dye-response curves determined for each reach. The initial approximation of the propane depth-averaged desorption coefficient (Kp') was determined by use ofthe following equation:

$$Kp' = \frac{24}{\bar{T}_d - \bar{T}_u} \ln \frac{C_u Q_u}{C_d Q_d}$$

where $\bar{T}$ is the travel time of a dye-tracer response curve centroid observed at a site, in hours since injection; C is the propane-gas plateau concentration, in micrograms per liter; Q is the stream discharge, in cubic meters per second; u and d are subscripts that designate the upstream and downstream sample sites, respectively; and 24 is a constant to convert hours to days. The approximation of Kp' by eq. 3 does not take into account longitudinal dispersion along a stream reach (Nobuhiro Yotsukura, U.S. Geological Survey, written commun., 1985). The use of the superposition principle states that the concentration plateau of a continuously-injected tracer can be calculated by adding the concentrations of subsequent slugs of tracer at a given time, based on theoretical response curves. This calculation is unaffected by longitudinal dispersion. The actual propane depth-averaged desorption coefficient (Kp) is determined through iterative solution of the equation:

$$\frac{C_u Q_u}{C_d Q_d} = \frac{\Sigma(C_{c,i,u} / A_u) \exp (-Kp\, T_{i,u}) \Delta T_{i,u}}{\Sigma(C_{c,i,d} / A_d) \exp (-Kp\, T_{i,d}) \Delta T_{i,d}} \qquad (4)$$

where T_i is the i^{th} hour since injection; $C_{c,i}$ is the dye concentration at T_i in micrograms per liter; and A is the area under the dye-response curve for the indicated sample site. The iterative process can easily be programmed on a computer and can use the Kp' approximated from eq. 3 as a starting point for the final determination of the propane tracer-gas desorption coefficient.

The depth-averaged reaeration coefficient (K_2) is related to the propane depth-averaged desorption coefficient (Kp) by the equation (Rathbun and others, 1978):

$$K_2 = (Kp\, 1.024^{(20^\circ - t)})\, 1.39 \qquad (5)$$

where t is the field water temperature in degrees Celsius.

RESULTS AND CONCLUSIONS

Tracer studies were conducted on August 20-21, 1985, September 17-19, 1985, and June 25-27, 1986, on the lower, middle, and upper reaches of Natty Pond Brook, respectively. The reaeration coefficients and diffusion constants shown in table 1 are the results of the dye- and gas-tracer analyses. The measured reaeration coefficients have a direct relation to the water-surface slopes for the reaches, as described in a state-wide analysis by Parker and Gay (1987). Oxygen-diffusion constants were calculated using eq. 2. The oxygen-productivity rates shown in table 1 were determined using the Stephens and Jennings (1976) program. The program totals the changes in oxygen concentration during the daytime after adjusting for thediffusion constant and respiration. Color, also shown in table 1, is an indicator of humic- and fulvic-acid concentration from drainage accrual. Highly colored water reduces light penetration through the water column, thereby reducing photosynthetic oxygen-production. Also, humic and fulvic acids can be a source of chemical oxygen demand within a river reach.

Table 1. Reach characteristics and results of tracer and productivity studies

Reach name	Wetland type	Water surface slope (m/m)	Color (Platinum cobalt units)	Reaeration coefficient (1/d)	Diffusion constant ($g/m^3/hr$)	Oxygen saturation (percent)	Net Daytime productivity ($g/m^3/d$)
Upper	scrub/shrub	.00004	20	0.45	0.17	105	+0.8
Middle	forested	.00500	25	8.45	3.3	92	-1.7
Lower	scrub/shrub	.00004	300	1.93	.75	29	-6.1

The difference in reaeration coefficients between reaches is related to the kinetic energy within the water column, as indicated by the water-surface slope. The upper and lower reaches had low water-surface slopes (0.00004 meters per meter) and correspondingly low reaeration coefficients (0.45 and 1.93 units per day, respectively). The higher reaeration coefficient in the lower reach may be the result of increased energy generated by the turning over of the water in the many sharp meanders, which are not present in the upper reach. The middle reach has a measured reaeration coefficient of 8.45 units per day, which is consistent with the higher energy level indicated by a water-surface slope of 0.005 meters per meter.

The decrease in oxygen-productivity rate from the upper reach to the lower reach of Natty Pond Brook can be related to the decrease in the water quality, as indicated by the increase in color. The water quality depends upon the percentage of water that is entering Natty Pond Brook directly from drainage accrual through the organic layers in the wetland areas. Stream reaches that have the largest ratio of wetland to total drainage also have the highest color values and the lowest apparent oxygen-productivity rates. The dissolved-oxygen saturation level decreased from 105 percent in the upper reach to 92 percent in the middle reach and to 29 percent in the lower reach, while color was 20, 25, and 300 platinum-cobalt units in the upper, middle, and lower reaches. The water-quality of the upper reach reflects upland drainage; the 2 percent of wetland area has minimal effect on the water-quality of the upper reach as indicated by a high percentage of oxygen saturation and a low color value. The upper reach has high sunlight exposure, which is common in an open low-sloped, scrub-shrub wetland, and has low-colored water (20 platinum cobalt units), which does not inhibit photosynthetic oxygen-production. As a result, the upper reach has an oxygen-productivity rate of +0.8 grams per cubic meter per day, supersaturated oxygen concentrations of 105 percent, and a low diffusion constant of 0.17 grams per cubic meter per hour. These conditions indicate that any oxygen demands are met during the daylight periods when periphyton and phytoplankton oxygen-production is occurring.

The middle reach is characterized by a relatively steep sloping wetland with mixed and forested species that do not allow much light to reach the water. The oxygen-productivity rate is reduced from that determined for the previous reach to -1.7 grams per cubic meter per day. The negative oxygen-productivity rate and reduced percent-oxygen saturation of 92 percent are probably the result of constant biological oxygen demand in a shaded, forested wetland reach that is somewhat compensated for by an increased diffusion constant of 3.3 grams per cubic meter per hour. The low color level of 25 platinum-cobalt units indicates that the humic- and fulvic-acid concentrations would notreduce light penetration and thereby inhibit photosynthetic oxygen-production. The lower reach is an open scrub-shrub, low-sloped wetland area that has a low diffusion constant of 1.93 grams per cubic meter per hour. The higher color value of 300 platinum-cobalt units in this reach reduces light penetration and photosynthetic oxygen-production during daylight periods; the high color also indicates the presence of humic and fulvic acids in levels associated with chemical oxygen demand. The color values increase from the middle reach to the lower reach, because the additional drainage accrual area represents the highest ratio of wetland area to total drainage area in the Natty Pond Brook basin. The low oxygen saturation of 29 percent and the oxygen-productivity rate of -6.1 grams per cubic meter per day for the lower reach implies that the biological oxygen demand indicated in the middle reach is significantly supplemented in the lower reach by a chemical oxygen demand contributed through drainage accrual.

The differences in apparent net oxygen-productivity rates among reaches result from differences in productivity, respiration, drainage accrual, and diffusion constant among the reaches. Each reach's productivity is highly related to drainage accrual, the dominant wetland

vegetative species that limits the amount of sunlight exposure to the water surface, and the color level that affects the light penetration within the water column. The effects of respiration within each reach is defined by the percentage of oxygen saturation and the dominant oxygen-demand processes indicated by color level. The decrease in the oxygen-productivity rates from +0.8 to -6.1 grams per cubic meter per day and the increase in color values from 20 to 300 platinum-cobalt units from the upper reach to the lower reach indicate a significant increase in chemical oxygen demand from drainage accrual. The diffusion constant for each reach was determined from reaeration coefficients measured using the steady-state, gas-tracer method. The use of measured reaeration coefficients in the place of estimated coefficients increased the confidence in all subsequent analyses of differences in oxygen-productivity rates among reaches.

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chapter seven

Wetlands and Stormwater

Wetlands and Stormwater Pollution Management

David F. Lakatos
Roy F. Weston, Inc.

Laurissa J. McNemar
Walter B. Satterthwaite Associates, Inc.

INTRODUCTION

Current emphasis in the management of urban water resouices is upon the control of storm runoff pollution. Recent state and local legislative and regulatory programs in many areas now require that stormwater pollution (otherwise known as nonpoint source runoff, NPS) be controlled. In response to this pressure, creative approaches are now being developed and implemented, some of which use existing and/or man-made wetlands in order to achieve storm pollution control goals.

The use of existing or constructed wetlands is proposed as an alternative for storage and treatment of stormwater runoff, in light of the cost and limited environmental advantages of more conventional stormwater methods. Highly complex ecosystems have provided degrees of treatment for urban and agricultural stormwater runoff comparable to that of conventional detention basins and even chemical treatment plants, according to completed studies on wetlands used for stormwater treatment (Harper, Wanielistga, Baker, Fries, and Livingston, 1985; Martin, 1985).

A stormwater management approach utilizing wetlands as water pollution control system "components" involves a degree of uncertainty, due to the complex and environmentally sensitive nature of wetland ecosystems. Various studies, though, do present favorable results with respect to the pollutant removal efficiencies of wetlands (Barten, 1983; Brown, 1984; Shih, 1981; Hickok, 1980; Hickok, Hannaman, and Wenck, 1977). However, because the use of wetlands for stormwater management purposes is relatively new, there are no handbooks or guidelines that discuss technical support data and present design criteria and procedures. This, coupled with the tremendous current interest in the stormwater use of such systems justifies a degree of caution in encouraging their use.

STORM RUNOFF POLLUTION

The need for storm runoff pollution control has long been recognized. The type and quality of pollutants carried by storm runoff, commonly resulting in "non-point source (NPS) pollution" of receiving waters, is highly variable (EPA, 1982). The pollutant characteristics of stormwater runoff are largely based on land use characteristics and vary with the duration and the intensity of a rainfall event (Metropolitan Washington Council of Governments, 1985). "Average" values for stormwater quality characteristics, however, have been generally defined in texts and references (EPA, 1980).

Table 1 illustrates some important points concerning NPS pollution, for selected pollutants, for example:

- The urbanization process generally increases the concentrations of NPS pollutants in almost all categories, with higher concentrations of most pollutants associated with higher degrees of development and impervious land surfaces.
- The construction process itself can be an important nonpoint source pollution generator, particularly for total suspended solids.
- Agricultural uses are also significant nonpoint source pollution generators, and any truly comprehensive nonpoint source management plan must address the characteristics and impacts of agricultural areas.
- Nonpoint source pollution is generally of "better" quality than raw wastewater for some pollutants (such as nutrients). However for other pollutants, nonpoint source pollution is comparable to treated wastewater effluent.

As was stated above, nonpoint source pollution is extremely variable and is dependent on many factors, including land use configuration and climate. The intensity and duration of a

rainfall event results in a variable pollution concentration in runoff, even over the actual period of runoff occurrence. That is, depending on soil characteristics, antecedent moisture (i.e., the preceding dry/wet weather) conditions, and land use types, the concentration of a particular pollutant can also vary with time during the actual rainfall event from a particular land surface area.

2. The time history of runoff from a site is represented by a "hydrograph", as Curve No. 2 illustrates. Typically the peak rate of runoff occurs sometime after the peak rainfall intensity.

(* These numbers are keyed to the numbers on the Figures, and each refers to a different type of curve on the Figures).

Table 1
Characteristics of Stormwater Runoff

Land Use	Pollutant Concentration (mg/L) Total-N	Total-P	TSS	Zinc	Lead	Iron
Forest	0.2	0.1	66	0	0	0.4
Agriculture	2.58	0.4	989	0	0	1.9
Construction Sites	4.0	0.5*	8,630	0	0	2.3
Commercial	VARIABLE					
- tourist	1.3	0.8*	4,020	0	0.5	4.2
- general	1.7	2.4	733	0.3	0.4	1.1
High Density Residential	2.5	0.4	249	0.17	0.15	1.4
Medium Density Residential	2.5	0.35	489	0.12	0.15	0.4
Industrial	VARIABLE RANGE - highly specific for type of industry. Comparable to commercial					
Recreation	0.6	0.4*	48	0	0	0.5
Open Space/Natural	1.35	0.06	8.7	0	0	0
Typical Urban Runoff (ranges) (20)	0.1-12	0.2-16	29-11,280			
Raw Wastewater (21)	35	10	420			
Treated Wastewater (21)**	20	7	50			

Source: Reference (10)

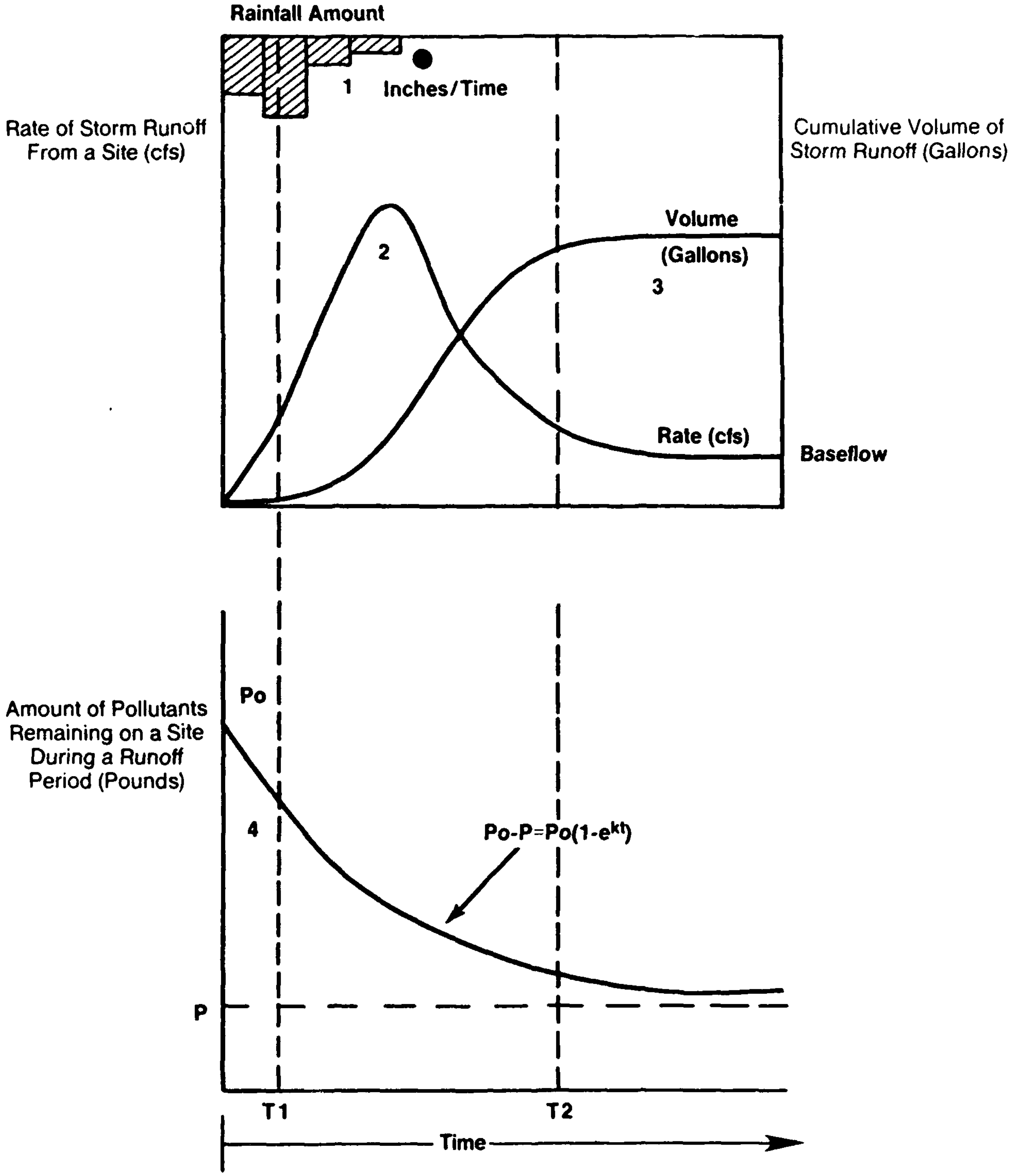
*Available data for Phosphate-P only
**For secondary treated wastewater

Testing (EPA, 1974) has shown, however, that the "worst" storm quality results from the initial "washing" of the land surface. This initial runoff pollutant concentration in stormwater is often referred to as the "first flush" effect and is illustrated in Figures 1A and 1B. The following information describes the various numbered notes on these figures:

*1. The characteristics of a rainfall event are represented by a "hyetograph", typically shown as in Curve No. 1 - which is a time history of rainfall for a site.

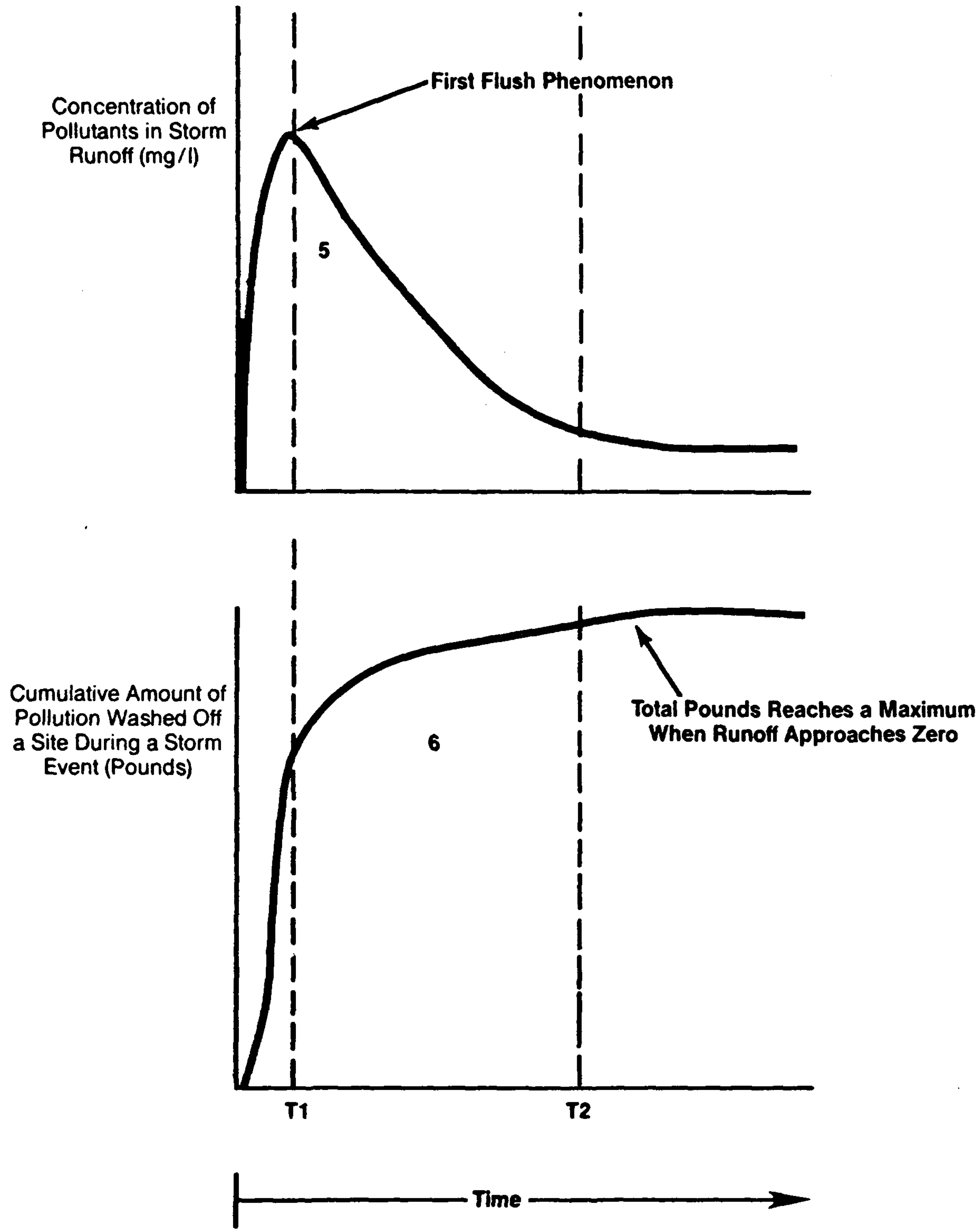
3. The cumulative volume curve, No. 3, represents the total amount of stormwater runoff from a site at a specified time. As the rate of runoff approaches its peak and then begins to decline (between T_1 and T_2), the total runoff volume rapidly increases, approaching its peak at T_2 when the runoff event approaches baseflow conditions.

4. On a pollutant Washoff curve, shown as No. 4, P_0 on the y axis represents the amount of a pollutant (in pounds) on the



NOTE: Each Curve, by Number, is Described in Accompanying Text.

FIGURE 1A FIRST FLUSH PHENOMENON



NOTE: Each Curve, by Number, is Described in Accompanying Text.

FIGURE 1B FIRST FLUSH PHENOMENON

site prior to a rainfall event. P represents the amount remaining after surface runoff has ceased. The curve represents the amount of pollutant washed off a site at any time T, and is dependent on rainfall or runoff excess, reflected in the constant "k."

5. Curve No. 5, sometimes called a "polluto-graphy", shows that the largest concentration of pollutants are washed off during T_1, where the rate and volume of runoff are low, preventing any significant dilution of the washoff pollutant quantity of Curve No. 4. The combination of rainfall timing, runoff and pollutant washoff results in the transport of typically high concentrations of pollutants to receiving waters in the beginning of a rainfall event. As the runoff event (see Curve No. 2) approaches T_2 the amount of pollutants remaining on a site approaches "P" (see Curve No. 4) and the pollutant concentration in the stormwater runoff approaches zero, due to the fact that less pollutants are available on the surface and larger volumes of runoff promote dilution. This phenomenon is referred to as the "First Flush Effect," where in simplistic terms, the dirtiest water leaves the site first.

6. Curve No. 6 illustrates the total amount of a pollutant that is transported from a site up to a specified time period. As an example, if a site stormwater collection basin existed, the total pounds of a pollutant measured in the basin at T_2 would be the result of the various mechanisms that take place in the rainfall-washoff process, as shown in curves 1 through 5, i.e., the combination of rainfall, runoff and land surface pollutant characteristics and the timing of these related events as they occur from the first drop of rainfall to the last drop of surface runoff.

WETLANDS AS STORMWATER POLLUTION MANAGEMENT FACILITIES

The current political and regulatory emphasis to provide quality control of stormwater runoff, during a time of tight financial conditions in the private land development industry, has resulted in many "creative" regulatory and technical/management approaches proposed by both the public and the private sectors. Wetland areas have been used not only because of their treatment effectiveness but also because of their other environmental amenities. Use of wetlands attempts to realize double benefits, i.e., preservation of important environmental areas while providing further water quality management functions--with an obvious goal of avoiding negative impacts to the important wetland areas in the process.

Functionally, wetlands perform differently than the typical detention basins. Detention basins operate primarily as "holding devices" to allow for large particulate pollutants in the storm runoff (along with whatever pollutants adhere to the particles) to settle to the bottom. The detention concept for stormwater quality management is enhanced with the use of "wet pond" detention basins, which involve a permanent pool of water in the bottom of the detention basin. This permanent pool of water helps to trap pollutants. A "wet pond" functions in a manner midway between a dry detention basin and a wetland stormwater quality control facility, relative to the number of mechanisms operating in each facility to improve water quality.

Wetlands, either natural or "engineered", can perform cleansing in addition to settling processes on the water which moves through them. The mechanisms by which wetlands remove nonpoint source pollutants include a combination of physical entrapment, chemical and microbial transformation, and biological utilization (EPA, 1980).

In light of the number of different mechanisms, it is easy to see how the effectiveness of wetland removal mechanisms can be highly variable and site specific. However, references are available which describe pollutant removal efficiency, reflecting the extensive work that has been done over the past decade with the use of wetlands for treated wastewater effluent polishing components (Boyt, Bayley, and Zoltek, 1977; Fetter, Soley, and Spangler, 1978; Gersberg et al, 1984; Nichols, 1983; Gersberg, Elkins, and Goldman, 1984).

Figure 2 illustrates, conceptually, the various mechanisms that work together to remove pollutants from water passing through a wetland area. Various pollution removal mechanisms that exist in a typical wetland include:

1. Sedimentation. This is one of the principal mechanisms for pollutant removal. Storm flows fill the available pool area and particulate matter settles. Vegetation slows the velocity of the incoming water, disperses the incoming water, and further enhances the settling/deposition process.

2. Adsorption. This is a physical and chemical process by which dissolved pollutants adhere to bottom sediments and, primarily, vegetation surfaces. Adsorption is a primary wetland pollutant removal mechanism, for both the more common pollutants (such as nutrients) as well as metals and even viruses.

3. Filtration. This occurs as particulates are mechanically filtered through sediments,

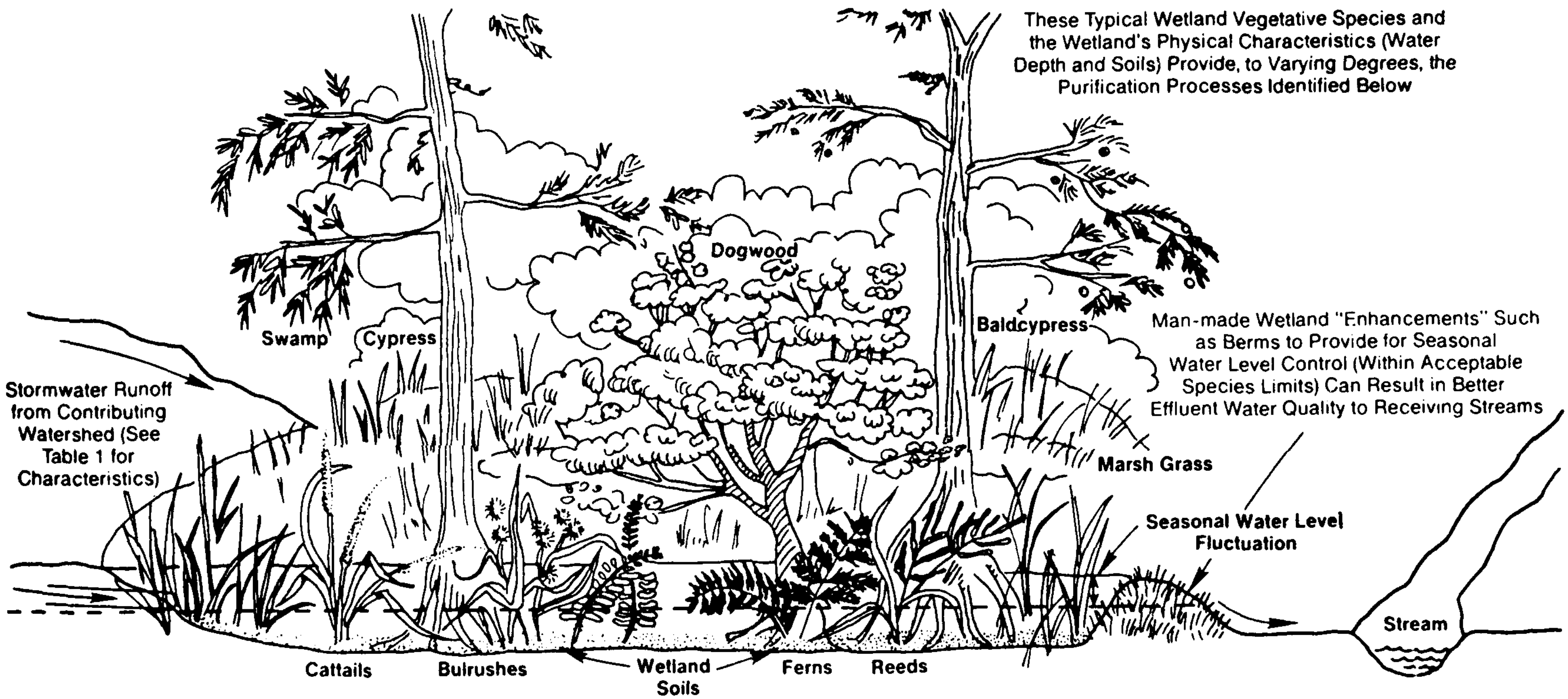


FIGURE 2 WETLANDS PURIFICATION PROCESSES

vegetation, and biota (as well as adsorption surface area). Filtration through vegetation and bottom soils can significantly reduce the migration of pollutants and bacteria along the length of the wetland.

4. Biological Assimilation. Wetland vegetation offers high pollutant absorption (biological uptake potential). It also provides an environment for significant microbial activity. Plants take up pollutants through their roots, which then allows further pollutant absorption within the plant tissues. Plants also absorb nutrients and ionic compounds from the water via shoots and leaves.

5. Microbial Decomposition. This occurs both aerobically and anaerobically in the water column, on the plant surfaces and within the soil. BOD removal in wetlands is carried out by decomposing micro-organisms. The bottom environment containing the wetland sediments is commonly anaerobic at the soil/water interface and provides the necessary conditions for the survival of denitrifying bacteria, which are a major part of the process for nitrogen removal. Heavy metals are converted to relatively insoluble sulfates in the reduced soils, which are characteristic of anaerobic conditions.

6. Chemical Decomposition. This mechanism involves photochemical reactions, chemical oxidation and reduction, and other processes.

A summary of actual performance results based on studies by many referenced authors is presented in Table 2. The results are for individual monitored events for the various referenced study areas. As such, they must be viewed as individual events and not broad general performance evaluation.

CURRENT OPINIONS ON WETLAND USE FOR STORMWATER MANAGEMENT

In order to provide a more representative summary of current trends and thinking based upon the opinions of a broad range of experienced individuals, a questionnaire was developed and distributed to a wide range of key individuals as part of a recent technical publication prepared by the authors of this paper (Lakatos and McNemar, 1986). The distribution list included experts in the field of stormwater management (both public and private sectors), as well as wetlands experts interested in stormwater management.

Many of the potential professionals were selected from the attendees of the Freshwater Wetlands and Wildlife Conference (1988) sponsored by the University of Georgia, Savannah River Ecology Laboratory. However, other experts in this emerging technical area were also contacted. The responses were coded and tallied statistically, so that similarities and differences in insight and opinions could be evaluated. The results of the evaluation of the questionnaires are presented below for four relevant categories.

Table 2
Performance Summary of Wetlands Studies for Pollutant Removal From Stormwater Runoff

References Study Site	Area of Wetland (Acres)	Pollutants Monitored	Concentrations Inflow (mg/L)	Outflow (mg/L)	Percent Pollutant Removal
Boney Marsh (18)	120	Total P	0.042	0.002	52.4
Armstrong Slough (18)	30	Total P	0.172	0.13	24.4
Ash Slough (18)	20	Total P	0.88	0.76	13.6
Chandler Slough (18)	938	Total P	0.31	0.22	29.0
Lake Josephine Wetland (19)	29	Total P	0.84	0.35	58.3
		TKN	4.71	2.78	41.0
		TSS	154.5	12.3	92.0
Jones Lake Wetland (19)	23	Total P	0.45	0.41	8.9
		TKN	3.79	3.07	19.0
		TSS	48.3	21.2	56.1
Lower Watkins Wetland (19)	106	Total P	2.39	2.17	9.2
		TKN	2.44	2.38	2.5
		TSS	128.3	33.8	73.7
Orlando, FL (2) combined detention pond and wetlands	1	Total Lead	1.47	0.09	93.9
		Sups Lead	1.40	0.06	95.7
		Diss Lead	0.08	0.03	62.5

WETLAND PERFORMANCE CONSIDERATION

Most respondents were apparently fully aware of the research summaries presented herein. Therefore, there was general agreement by almost all respondents that wetlands could, in fact, perform important water quality management functions. However, numerous performance concerns were identified in the questionnaire responses.

Despite these concerns, over 78 percent of respondents were in favor of the use of wetlands for stormwater quality control. In contrast, only 5 percent were not in favor and 17 percent were essentially neutral to the idea. Further, of those respondents who were in favor of the use of wetlands systems for stormwater quality control, over 61 percent felt this way because of the effectiveness of wetlands for pollutant removal. However, 14 percent of the total number of respondents, primarily those who were not in favor of the use of wetlands for stormwater quality control, felt that not enough was yet known to allow for the broader use and application of wetland stormwater facilities. Another important insight from the questionnaires was that, of those who were in favor of the use of wetlands for stormwater quality control, 82 percent support reclamation or creation of new wetland areas, while 18 percent supported only the creation of new wetland areas.

In addition to the above, and related to the performance of wetlands stormwater management facilities, the following statistics were identified concerning the response to the advantages of the use of wetlands for stormwater quality control:

- 21 percent highlighted the preservation of existing and creation of new wetland areas.
- 3 percent identified wildlife protection and enhancement as an important advantage advantage of using wetlands for stormwater management.
- 24 percent identified pollutant removal effectiveness for downstream water quality as the most important advantage that wetland systems may have for stormwater management.

DESIGN CONSIDERATIONS

The topic of proper design guidelines was one that was addressed by many questionnaire respondents, with a majority of respondents (64 percent) calling for research to be conducted in order to develop effective design guidelines. Thirty-six percent of the respondents felt that adequate research has been done to allow the development of design guidelines and to justify promoting the use of wetlands as stormwater management facilities. Ninety-two percent of these respondents are involved in or have completed research on applications of wetlands for stormwater quality control.

Some of the statistical results corresponding to the above comments are presented below.

- 4 percent highlighted the need for a forebay and/or a sedimentation basin ahead of the wetland treatment area to minimize the sediment load to the wetland.
- 17 percent recommended the development of specific effluent quality standards.
- 13 percent recommended the development of hydraulic design criteria as it relates to retention time in the wetland.
- 9 percent felt that design and/or performance guidelines should specify vegetation types for the wetlands treatment system.
- 57 percent of respondents generally recommended <u>all</u> of the above types of design and/or performance criteria for an eventual set of engineering guidelines.

MAINTENANCE CONSIDERATIONS FOR WETLAND STORMWATER MANAGEMENT SYSTEMS

Almost all of the respondents highlighted the need for maintenance of wetlands stormwater management systems, especially when they are utilized for a controlled design purpose. Of the total number of questionnaire respondents that answered the maintenance related questions, the following maintenance precautions were identified (these percentages overlap as some respondents identified more than one of the following):

- 41 percent identified the removal and disposal of sediment as a primary long-term maintenance item.
- 10 percent identified the need for a contingency plan to be developed during initial design, in the event that the wetland treatment system does not perform properly, or stops performing effectively.
- 38 percent highlighted the need for long-term monitoring of overall pollution control effectiveness, as a legitimate maintenance concern.
- 17 percent identified water level management as a primary long-term maintenance consideration.
- 28 percent identified vegetation monitor-

ing as being a primary long-term maintenance item.

- 7 percent identified the fact that effective design, with specific provisions to minimize maintenance issues and/or concerns, is a primary component for the design of wetland stormwater management systems.

RESEARCH NEEDS FOR WETLAND STORMWATER MANAGEMENT SYSTEMS

All of the questionnaire respondents, whether or not in favor of wetland stormwater management systems, expressed the need for continued research. The highest priority research need was to determine the long-term impacts on the sensitive wetland environmental system itself. Additionally, numerous respondents identified the need for continued research in the areas of system performance, design criteria and maintenance requirements.

SUMMARY

In the opinion of the authors of this paper, the final consensus of the experts contacted for this paper was that wetland systems may be an extremely important part of a water quality management effort for an area. Their ability to effectively "cleanse" storm runoff is well documented and generally accepted. However, there are also major concerns about the misuse of these systems. This appeared to be especially true if wetland use grows significantly to the point that many are designed by individuals who do not have the capability and the experience (or necessary sensitivity to environmental importance) to do so in an acceptable fashion.

Many experts feel that, despite potential problems, the emphasis on wetland use and/or enhancement may help to promote a higher level of concern for wetland areas. Some feel that the benefits of new wetland area development, particularly in urban areas, outweigh the potential for environmental problems. All experts, however, point to a need to move very cautiously in the beginning stages of wetland system use, paying particularly close attention to at least those types of problems that we know can occur. In addition, the need for more research and testing/evaluation was emphasized, as well as monitoring of demonstration-type sites. Despite the potential for problems, the use of wetland systems for stormwater quality management was generally endorsed.

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The Creation and Use of Wetlands for Stormwater Treatment

Chris Athanas
Horn Point Laboratories
University of Maryland

Earl Shaver
Maryland Department of the Environment

INTRODUCTION

The pattern of stormwater runoff changes greatly as a watershed is developed. The increase in such impervious surfaces as parking lots and roads results in a greater volume of surface runoff leaving the watershed immediately after a storm. This rapid runoff rate can result in downstream erosion.

The most basic treatment of stormwater runoff involves retarding the rate of runoff. A common method of doing this is to impound the runoff so that the runoff leaves the basin at a more acceptable rate, which often corresponds to some pre-development rate. This type of treatment should result in a decrease in erosion downstream.

One of the drawbacks to this type of treatment is that stormwater runoff is detained for only several hours in the basin. This is not enough time for particulate material to settle out. To increase the amount of time available for sedimentation, outlets have been made smaller in many basins. This results in extended periods of detention, often as long as 24 hours. The use of extended detention basins significantly increases the removal of suspended solids from stormwater runoff (MWCOG, 1983). Extended detention, however, appears to have little effect on the dissolved nutrient component of stormwater runoff (MWCOG, 1983). These nutrients do not settle out and must be removed biologically.

LAKES AND DEEP PONDS FOR STORMWATER TREATMENT

The use of long term ponds and lakes is one method for allowing biological processes to remove dissolved nutrients. With this approach, a certain amount of runoff is permanently retained in the lake or pond water. The larger the body of water, the larger the volume of runoff that can be retained. Detention times vary, but in most cases new stormwater runoff at the next runoff event will displace an equal volume of lake water. Although the increased detention times that result from this method are important to sedimentation, the biological processes in a permanent body of water are probably equally important.

The best data set on the treatment of stormwater runoff using lakes and deep ponds was gathered during the Environmental Protection Agency's Nationwide Urban Runoff Program (NURP) (U.S. E.P.A., 1983). Most of the sites studied were deeper bodies of water, with depths greater than about 3 feet.

Figure 1, based on the NURP data, shows the removal of total nitrogen and phosphorus from stormwater runoff by lakes and deep ponds. Nitrogen and phosphorus removals are plotted against the ratio of basin surface area to catchment surface area, expressed as a percentage. None of the basins had a pond surface area that was larger than 3% of the catchment area.

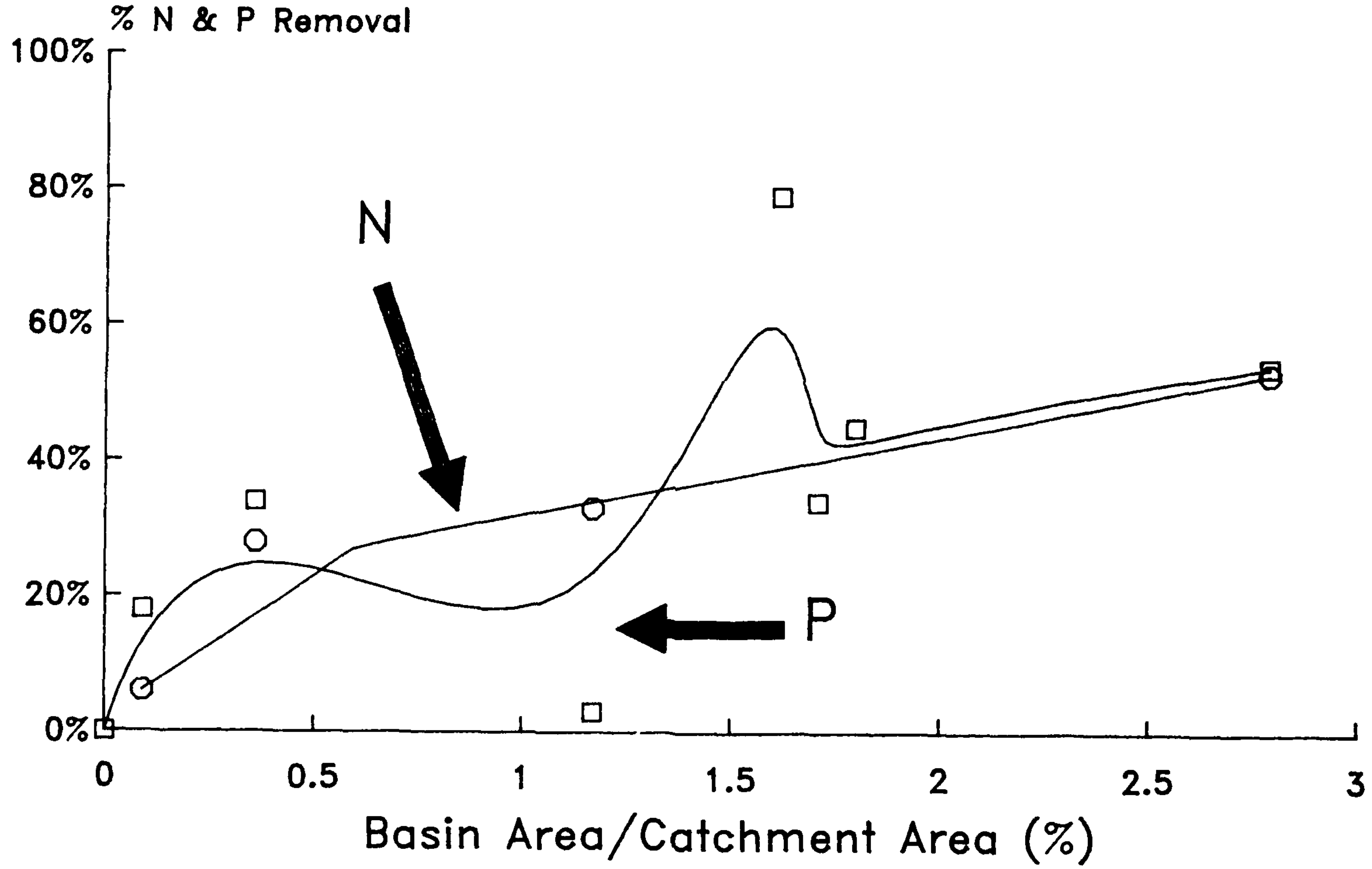
For nitrogen, there is a steady increase in removal rate up to about 50% for a pond area that is about 3% of the area of the catchment. The curve for phosphorus is similar to that of nitrogen, with a removal rate of around 50% when the pond area is 3% of the catchment area.

Another way to look at the NURP data is to plot the removal rates for nitrogen and phosphorus against a volume ratio. This is done in figure 2, which shows nitrogen and phosphorus removal rates plotted against the ratio of basin volume to the average runoff volume. Values are expressed as ratios. Percent removals of nitrogen and phosphorus have similar curves in figure 2. At ratios between 1 and 2 (which means a pond volume equal to or twice as great as the runoff volume), most removal rates were around 30 or 35%. Although the data is sparse for larger basins, the plot suggests that for removals of more than 50%, the basin volume had to be more than 3 or 4 times the runoff volume.

THE USE OF WETLANDS FOR TREATING STORMWATER RUNOFF

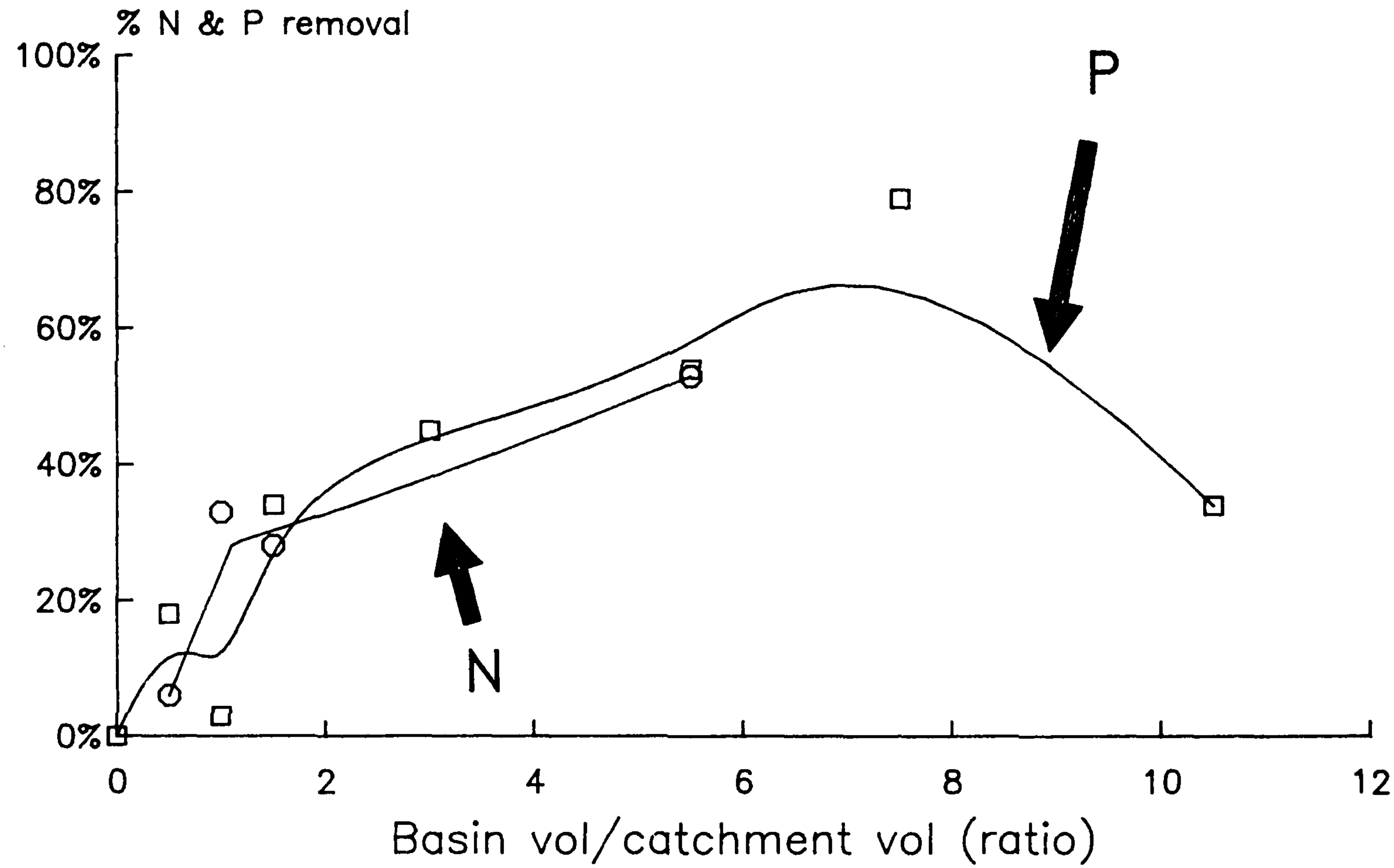
With the data from the EPA's Urban Runoff Project indicating that permanent bodies of water could improve the quality of stormwater runoff,

Figure 1. P,N Removal
Surface Area Ratios



Date from EPA 1983

Figure 2. P,N Removal
Volume Ratios



Data from EPA 1983

interest has increased in the potential use of wetlands for treating stormwater runoff. Much of this interest arises from the recognition that wetlands, aside from their possible pollution removing potential, can be valuable additions to the urban landscape. Urban wetlands can provide wildlife habitat, educational opportunities, and park environments for the surrounding community.

Although shallow wetlands may not have the long-term storage capacity of lakes and deep ponds, the potential for high rates of biological activity in wetlands may compensate for the lack of volume. Shallow areas permit the growth of emergent and submerged vegetation, benthis algae, and bacteria, and may result in higher water temperatures and greater sunlight intensities -- factors which promote biological processes.

Figure 3, based on data from Wetzel (1975), shows the difference between bacterial activity in the emergent vegetation portion of a lake and in the deeper, pelagic portion. Summer activity in the shallow area was higher than in the pelagic portion, and higher than at any other time during the year.

Biological processes in the wetland act on nutrients, phosphorus and other materials in stormwater runoff by transforming them into plant material and other less troublesome substances. In general, we can say that the major inorganic nutrients entering the wetland, including nitrate, ammonia, and phosphate, are immediately available for uptake by plants, bacteria, and fungi. One of the purposes of wetland stormwater basins is to encourage such growth where we want it, instead of downstream in a reservoir or some other body of water.

These biological processes, plus sedimentation, result in the removal and transformation of nutrients by wetlands. Unfortunately, at the present time there is not much data on the removal of nutrients from stormwater runoff by wetlands. However, some interesting data on the removal, by wetlands, of nutrients from sewage effluent and related sources are available.

Nichols, in 1983, summarized from the literature the percent removal of nitrogen and phosphorus from sewage effluent and fertilizer inputs from a very diverse group of wetlands, including cattail marshes, a Florida hardwood swamp, an Irish blanket bog, and others. They were all freshwater, non-tidal wetlands. He found that when total nitrogen inputs were kept below approximately 100 kg/ha of wetland per year, removal efficiencies were greater than 70%. Similarly, when total phosphorus inputs were kept below approximately 15 kg/ha of wetland per year, removal efficiencies were about 68%. Removal efficiencies declined rapidly at higher nutrient loadings.

The application of nitrogen and phosphorus to the high efficiency wetlands reported on by Nichols (1983) was done in a controlled manner, with hydraulic loading kept low. In addition, wastewater inputs can be made during times of low rainfall to increase the retention time of the wastewater in the wetlands. This results in the increased exposure of the nutrients to the biological processes of the wetland.

In contrast, stormwater basins receive their nutrient loads during periods of rainfall. The loading rates discussed by Nichols (1983) can be compared to the loading received by a hypothetical wetland used for stormwater treatment. This wetland will have a surface area of 0.4 ha, a watershed of 20 ha, a runoff coefficient of 0.5 (half the rainfall received is converted to runoff), and an annual rainfall of 100 cm. Total nitrogen and phosphorus loads are calculated using midpoint values of 2.8 mg/l (N) and 0.4 mg/l (P) from the EPA's ranges of values for event mean concentrations of these nutrients from stormwater runoff (U.S. E.P.A. 1983).

The nutrient loads to this hypothetical wetland are calculated to be more than 700 kg of nitrogen per hectare of wetland, and more than 100 kg of phosphorus per hectare of wetland. Based on figures presented by Nichols (1983), these loadings would result in removals of approximately 40-50% for phosphorus, and approximately 30-40% for nitrogen. These values are somewhat lower than those shown in figures 1 and 2 for the NURP data.

STORMWATER TREATMENT IN MARYLAND

The state of Maryland has implemented an aggressive stormwater management program whose main goal is to maintain pre-development runoff characteristics of a watershed after development has occurred. Specifically, storms of 2 and 10 year frequencies may not create peak discharges above their pre-development levels. As discussed previously, this usually requires the construction of a dry detention basin to retard the rate of runoff and reduce it to pre-development levels. Although the runoff volume of these streams is increased, the additional volume is stored in the basin and released over a longer time period so that the peak discharge is not increased.

In order to improve water quality, the state of Maryland is requiring that stormwater basins having a shallow, permanent pool of water be utilized whenever possible. Although all of the evidence regarding the benefit of wetlands for treating stormwater runoff is not yet in, several factors influenced this decision. First, field reviews of existing dry detention ponds conducted throughout the state found that more than half of the 389 dry ponds were becoming or had already become, wetlands.

This reversion to wetland was primarily caused by blocked outlets. Thus, it was decided to include properly designed wetlands in many future basin projects. Second, there appeared to be sufficient information available to design wetland basins that would contribute to water quality, wildlife habitat, and community esthetics.

Aware that more research was needed on the use of artificial wetlands for treating stormwater runoff, the Maryland Sediment and Stormwater Division and the University of Maryland implemented a cooperative agreement for research on the problem, which focused on three sites in Maryland.

RESEARCH

The Mays Chapel Site

The Mays Chapel wetland stormwater basin is located in the watershed of one of the principal reservoirs of Baltimore, Maryland. This basin was converted into a shallow wetland basin from a dry stormwater basin in July of 1985. The University of Maryland established and is studying the vegetation in the wetland. Although the City of Baltimore is monitoring the water quality effects of the basin, this data is not yet available.

The basin was graded so that approximately 2/3 of the wetland would be 30 cm deep or less. The remaining third ranges from 30 to 60 cm deep. The wetland has a surface area of 0.3 ha and the area draining into it has an area of about 40 ha. The basin outlet is constructed so that the runoff from a 1/2 year storm is detained in the basin for up to 14 hours.

Three wetland species were obtained from a local nursery and planted on the wetland. These included Sagittaria latifolia (duck potato), Scirpus americanus (common three-square), and Leersia oryzoides (rice cutgrass). All of these species spread aggressively, a trait valuable when establishing a wetland. A total of 2,600 individuals of S. latifolia, 375 S. americanus, and 375 L. oryzoides were established. Approximately 50% of the wetland area was planted.

The vegetation on the wetland has been sampled since 1985 in order to estimate peak biomass and diversity of the wetland vegetative community. For sampling purposes the wetland was divided into 2 areas, the first including that part of the wetland within 4 meters of the shoreline, and the second including the rest of the planted areas. The water depth in the first area ranged, on a constant slope, from less than a centimeter to about 30 cm. In the second area the depth was approximately 30 to 40 cm.

Figure 4 shows the peak standing crops in those two zones from 1985 to 1987. Peak standing crops increased greatly in 1986 and 1987, with the near shore zone showing higher values than the deeper portion of the wetland. Visually, the near shore area has greater coverage than the deeper area, which has become noticeably clumped. Greater plant densities with decreasing water depth has also been found by other researchers (Fulton 1983; Pierce 1983; Pierce et al., 1983).

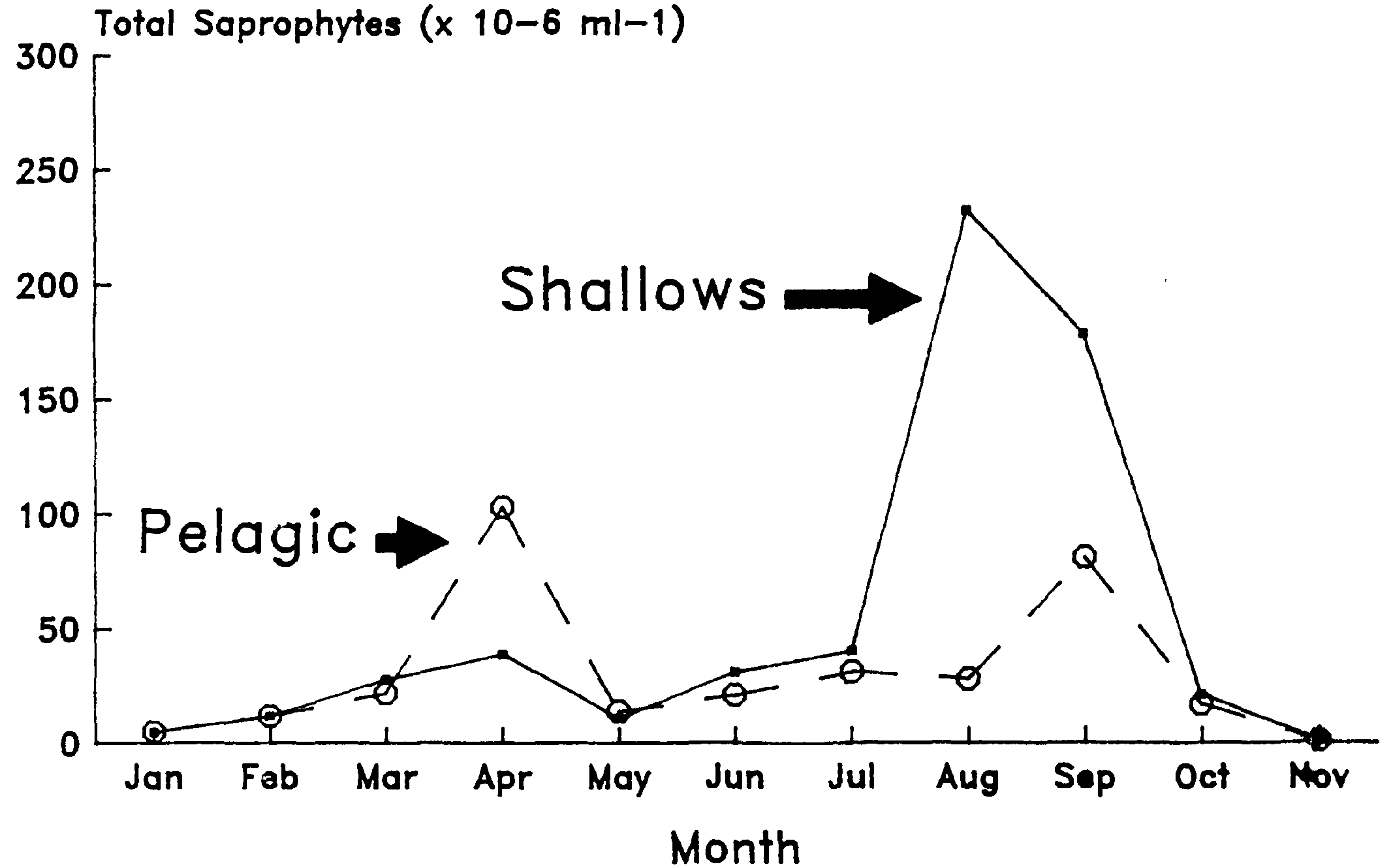
A shallow wetland pond left alone will almost certainly develop some vegetation over time. What is not clear is the time frame for such development and the species composition of the community. At Mays Chapel, two years after the initial planting, only 2 unplanted species, Alisma plantago-aquatica (water plantain) and Eleocharis sp. (spike rush) were present in any appreciable numbers. Abundance is very low, however, with biomass less than 1 g/m^2. A third species, Scirpus validus (soft-stem bulrush) is present in several isolated clumps. At this time artificial establishment has easily been the controlling factor in the vegetated state of the wetland.

The Prince Georges and Queen Ann Sites

This year we have begun research on two additional artificial wetlands. The first of these wetlands, the Prince Georges site, is a 0.4 ha wetland which receives runoff from a 40 ha commercial watershed. This site is located in Maryland approximately 12 miles from Washington, D.C. The wetland was converted from a dry basin during the summer of 1987. The wetland is shallow, with a maximum depth of 30 cm. No vegetation has been planted on this wetland because the research program includes monitoring the natural development of the vegetative community. Stormwater inputs and outputs from the basin are being monitored using Palmer-Bowlus flumes and automatic flow monitoring equipment. Flow-proportional water samples are also being taken automatically for determination of suspended solids and nutrient concentrations.

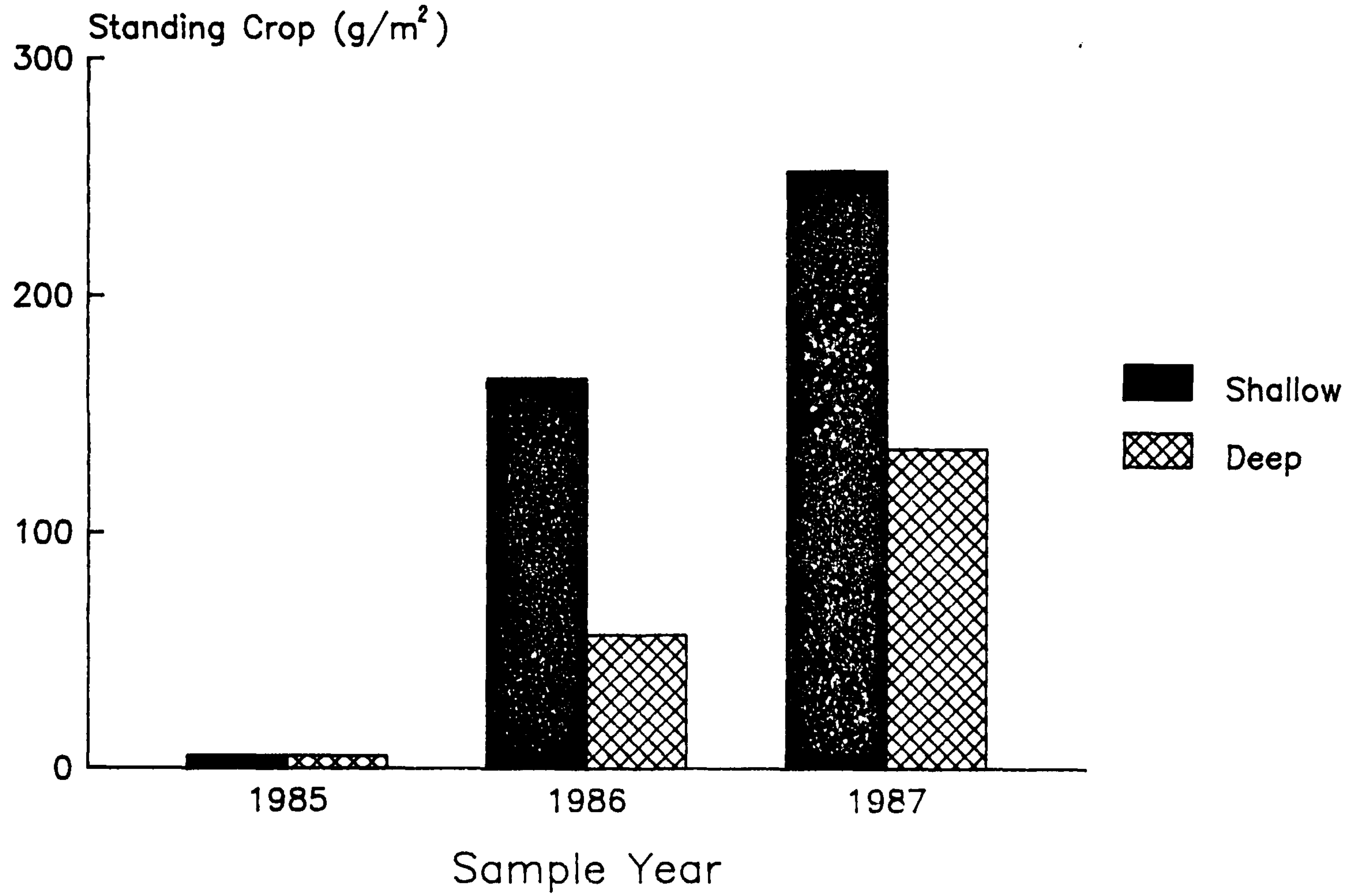
The second research site begun this year is the Queen Ann site, a 0.24 ha wetland located on the eastern shore of the Chesapeake Bay. Runoff to this site is from the 6.4 ha grounds of a local high school. This wetland has an average depth of 30 cm, although the deepest depth is 60 cm. 0.99 m H-flumes are being used to monitor flows at one input point and the single output point, while the second input point is a calibrated pipe. As at the Prince Georges wetland, flow proportional water samples are taken for suspended solids and nutrient determinations. Three species of plants are being established in this wetland. They include Scirpus americanus (common three square), Saururus cernuus (lizards tail), and Sagittaria latifolia (duck potato). A total of approximately 4000 plants will be planted in this wetland.

Figure 3. Saprophyte Numbers
Shallow vegetated vs. Pelagic



Adapted from Wetzel 1975

Figure 4. Peak biomass (g/m^2) of wetland emergents.



RESEARCH GOALS

There are two primary goals involved in this research. The first is to determine the effectiveness of small, artificial wetlands of a particular design in removing pollutants from stormwater runoff. This basic information will tell us if such wetlands are useful in pollution removal, and if design modifications are necessary.

The second research goal is to evaluate the role of wetland vegetation in pollution control by comparing the development of the vegetation over time with changes in pollution removal. Changes in the wetland vegetation, including species composition, plant structure, and plant density, are probably the most easily manipulated of all basin characteristics. Artificially induced changes in vegetation could be effective methods for improving the control of pollution.

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Use of Wetlands in Stormwater Management: A Case Example of Local Government Involvement

Peggy B. Johnson
Clinton River Watershed Council

INTRODUCTION

This paper will address the use and protection of wetlands in the context of stormwater management, focusing on the role of local governments. This paper presents guidelines for using wetlands for stormwater management gleaned from a literature search and consultations with national and Michigan experts. These have been accepted by local policy planners.

THE SETTING: OVERVIEW

The setting is the Clinton River Watershed, immediately north of the City of Detroit. Originally there were extensive swamplands here where major cities now exist. Today Michigan's wetlands are estimated to consist of only one third of their original acreage. There has been an even greater loss of wetlands in the Detroit Metropolitan area.

Nearly all of the wetlands of southeast Michigan are groundwater discharge rather than recharge areas. This means that groundwater contamination through use of wetlands is not a major concern. However, this must be verified for specific wetlands by field investigations.

In communities in the headwaters area, the extent of wetlands loss runs around 85%. Studies have shown that these wetlands play key roles in Clinton River flood peak mitigation. Much of the flood damage in Michigan occurs on the Clinton.

Point source discharges have been successfully controlled in this area. However, non-point source pollutants associated with urban runoff dominate the stretches of the river where water quality standards are not met. Corps of Engineers flood control planning had focused attention on the need for stormwater runoff control consistent with future development in the watershed.

THE PROJECT

The Clinton River Watershed Council, a voluntary association of local governments formed under the Michigan Local River Management Act, sought federal funding through the Michigan Department of Natural Resources for a demonstration project to undertake an assessment of the state of stormwater management in the river basin and to work toward improved management with three pilot communities that share the Paint Creek subwatershed. The purpose of the demonstration project was to compile state-of-the-art information from stormwater management experiences around the nation and explore, with local planning communities, the policies, programs, and financing mechanisms appropriate to the circumstances in each community.

The project objectives were to: develop master stormwater management policy plans in each of the three pilot communities; (2) place the planning in the context of the overall watershed perspectives; (3) provide implementation tools such as ordinances, financing options, and a compilation of best management practices; and (4) produce a guidebook on local stormwater management for communities across Michigan.

THE STORMWATER MANAGEMENT PROBLEM

Without careful planning as a community changes from rural to suburban to urban, severe stormwater runoff problems can occur: increased flooding, degraded water quality, and unstable stream channels.

In the past, stormwater was considered to be "the enemy". Manmade drains were built to hasten the water to the nearest lake or stream and out of the local community. In many cases, this simply transferred the problem downstream. Repeated enlargement of municipal drain systems through new public works was very costly. Wetlands were viewed as wastelands to be converted into useable dry lands by draining and filling.

Stormwater management proceeded from an early emphasis on drainage, with piecemeal projects and no planning or management, to the beginning of on-site detention in the late 1960's. Today the trend is towards watershed management and facilities which combine quantity and quality control through the use of selected storage areas.

By planning for the stormwater system ahead of community growth, control measures can be

implemented at the same time as new development occurs.

WETLANDS IN THE CONTEXT OF LOCAL GOVERNMENT STORMWATER MANAGEMENT PLANNING

Need for Local Government Involvement

The shift from a remedial to a planned approach for stormwater management hinges on the involvement of local governments and their ability to review proposed land uses and to finance public projects for land acquisition or construction. The understanding and cooperation of developers and residents is crucial.

A single Michigan law which delineates stormwater management responsibilities and establishes uniform standards does not exist. Instead an array of state laws/enabling statutes provide local governments with tools for managing stormwater. The patchwork of activities by state and county agencies are helpful, but only local governments can set and carry out stormwater management policies in a logical and comprehensive fashion.

Local stormwater management needs vary depending on such factors as the location in the watershed; extent of existing development and current problems; future land use expectation; configuration of water resources, soils, and topography; administrative and financing capabilities of local government; and preferences of residents. No one "canned plan" is appropriate to the varying local circumstances.

In one of the project's pilot communities there are many lakes and wetlands, flat topography, and the headwaters of several streams. The lakes provide ample storage for stormwater although there is concern for shoreland flooding. With extensive lakeside residential development there is also a high concern for the protection of lake water quality and the role of wetlands as buffers. In contrast, another township has few lakes but two major creeks with steep valleys. Paint Creek is the major remaining trout stream in the Detroit area. Here there is interest in routing stormwater through strategically located wetlands to protect creek water quality.

Local planning for the stormwater system addresses the basic questions: Where did the runoff originate? By what pathways will it be transported? In what places will stormwater be stored in the community? What is the receiving outlet. Natural wetlands, modified wetlands, or created wetlands (wet pond detention basins) can be key storage areas. But, special consideration must be given to wetlands that are a receiving waterbody without outlet (retention area).

Watershed Planning

Small watersheds within a municipality are the building blocks for planning the municipal system and the context for reviewing site plans. Municipal level planning objectives must be mindful of the larger watershed context and coordinated with other municipalities.

Typically a community will progress through successive phases of local stormwater management. In the early stages of community growth the emphasis will be on requirements for developers for on-site stormwater management measures. Off-site impacts from new developments can be minimized by obtaining information about the site early in the site planning and review process, its existing and proposed drainage, plus the small watershed within which it lies. Next, the local government can undertake analysis of small watersheds where development is imminent to delineate the management options and identify opportunities to optimize a developer's contribution towards construction of the local stormwater system. This may facilitate watershed storage areas shared by more than a single development, such as strategically located wetlands. Finally, when new developments are scattered throughout the community, a stormwater master plan including all municipal watersheds may be completed. By then it is likely that the local government will have the capacity to finance some public works, provide municipal maintenance of the stormwater system, and perhaps establish utility fees for tap-ins and flow contributions to the municipal system.

Local Policy Conclusions

From a public standpoint, the use of wetlands for stormwater management offers both advantages and disadvantages.

Advocates of using wetlands for stormwater management point out the following:

- Wetlands are "expensive" features of a development site, since the area cannot be filled and used for buildings. By using wetlands for stormwater detention, a "positive" use occurs, and pressures to fill wetlands are reduced.
- Wetlands often need to receive some runoff in order to sustain vegetation and wetland conditions. Diversion of stormwater away from wetlands may dry up the wetland.
- Wetlands have different natural values, reflecting different types of vegetation, size, hydrology, location, and extent of human impact. The "low value" wetlands may actually be enhanced if they are used for stormwater management. For example,

dredging out an open pond in homogenous emergent vegetation wetland can benefit sedimentation functions and wildlife habitat diversity.

Advocates of wetlands protection (without stormwater detention) point out the following:

- Urban stormwater runoff can carry high volumes of sediment and pollutants which do not benefit wetlands wildlife and water quality.
- It is not essential to use naturally occurring wetlands for stormwater detention, since man-made wet ponds and basins can be designed to provide equivalent quantity and quality control.
- Public and private stormwater facility maintenance programs are not well developed. Use of wetlands for stormwater without maintenance will lead to excessive wetlands destruction.

A local government can balance these two viewpoints by completing the following steps:

1. Identify all high value wetlands, and wetlands which are contiguous to lakes and streams. High value wetlands should not become stormwater management areas, and ecological studies in the community should identify such locations.

2. Undertake "small" watershed planning to identify possible locations where isolated wetlands may be useful for stormwater detention.

3. Develop strong local government review standards for erosion control. Many potential problems with stormwater and wetlands can be avoided if effective erosion control measures are in place throughout the time that the small watershed is undergoing development. Effective measures will usually necessitate diversion structures and sediment traps, not just hay bale berms.

4. Ensure that local officials have accurate information about wetland functions and locations at the time that development proposals are reviewed. Although available wetlands maps provide useful general information, site investigation is needed to properly identify wetland locations and values. The local government perspective on "wetland values" is usually different from the property owner's view, since the property owner wishes to develop his/her land. Local governments need their own wetlands consultants if substantial wetlands are located on the development site.

It is sound public policy to use but not overuse wetlands. Since the long term impacts of urban stormwater on various types of wetlands or the watershed ecology are not well understood, proceeding with caution is warranted. Those wetlands with high wildlife habitat and ecological value should be avoided; they should be preserved. Other wetlands may be utilized based on an understanding of their functions and values, and likely impacts on them. It is best to manage wetlands to optimize several important functions rather than any single function. For example, a single large wetland in a watershed might satisfy the flood water detention requirements; but multiple smaller wetlands would be better for preventing water quality degradation and channel erosion.

GUIDELINES FOR USING WETLANDS FOR STORMWATER MANAGEMENT

Defining Wetland Boundaries

In the Michigan Wetland Protection Act, wetlands are defined as "land characterized by the presence of water at a frequency and duration sufficient to support, and under normal circumstances do support wetland vegetation or aquatic life." In other words, vegetation is referenced as the key indicator. While this definition may be satisfactory from a science perspective, it does not directly lead to easy identification of wetlands in the field. Knowledgeable wetlands experts rely on field investigations to identify wetlands locations, usually combining vegetative indicators with soil indicators.

Wetlands maps from the U.S. Fish and Wildlife Service (National Wetlands Inventory maps prepared in 1980) are useful as general indicators of wetlands. The National Wetlands inventory maps are usually more accurate and useful than either U.S. Geological Survey quadrangle maps or general soil survey maps. However, these maps were prepared from aerial photographs - not field surveys. For field survey work, however, it is best to use all available maps to identify potential wetland locations and to confirm boundaries through field inspections.

The boundary between wetlands and uplands is rarely a distinct line, a factor which makes field investigation even more important. A transition zone is often present where wetland species of plants are mixed with upland types. Sometimes soils can be a very useful indicator, since the new hydric soils classification of the U.S. Soil Conservation Service reflects wetlands ecosystems.

Although the transition zone is part of a wetland, it may become an "area of negotiation" between the developer and public agencies. For sound wetland management, it is prudent to protect this transition zone as a buffer between the upland and open water area of the wetland.

Using Wetlands as Primary Stormwater Detention Areas

Proposals to use the wetlands as the primary stormwater detention areas for developments must be critically reviewed. One of the most basic questions is erosion control. Will the wetlands receive sediment-laden stormwater, or will sediment traps in uplands areas be used?

Sediment-laden runoff should only be allowed in wetlands where periodic dredging and maintenance can be carried out (similar to a wet detention basin). Alternatively and preferably, permanent upland sediment traps near the wetland boundary should be provided. There is a clear gap in Michigan laws: if one proposes to dump a truck load of dirt into a wetland a fill permit is required; but the dirt may be slurried into a wetland via a stormwater discharge and no permit is needed.

An approach to stormwater detention which is sometimes possible is dredging to enlarge an existing wetland. State-local government coordination is essential in such cases to assure that state wetlands permit requirements as well as local stormwater management and wetlands protection requirements are met.

Wetlands used for primary stormwater detention should be set back 100 feet or more from any lake or stream. The setback assures a reasonable buffer between stormwater and the lake or stream, allowing nutrient and sediment removal to function.

Even a 100-foot setback will not be sufficient, however, if stormwater is piped or otherwise discharged into a single channel in the wetland. Stormwater should be dispersed over the ground or directed through smaller pipes to several areas of the wetland. Without careful design of the stormwater discharge to the wetland, a channel will be created which simply connects stormwater directly to the lake or stream.

Shared Use of Wetlands

Wetlands boundaries rarely respect property lines; a wetland on a development site will often cross into adjacent properties.

Whenever a developer proposes to use wetlands in such a way that the quality or quantity of stormwater runoff to a wetlands is different than the runoff prior to the development, a wetlands use easement should be obtained. Without an easement, the developer (and future landowners) may be subject to lawsuits from adjacent wetland owners. The need for easements is particularly important if several property owners propose to use the same wetland for stormwater retention purposes.

The major challenges with shared wetland use relate to financing and management. Which landowner should be responsible for wetland improvements? How can maintenance responsibilities be fairly shared? What happens if one property owner is ready to develop before the others.

Local governments who wish to encourage the use of shared wetlands for stormwater management may need to investigate the possibility of purchasing use options in wetlands at pre-development costs, or providing other incentives to support landowner cooperation.

After stormwater has been temporarily stored in a detention basin on-site, it must be released to a drain, creek, lake, or other wetland. Even though stormwater velocities and release rates are reduced by on-site detention, the total volume of stormwater discharged from the site remains significant. Urban development increases the volume of stormwater leaving the site - whether or not on-site detention is possible. When release is to a wetland without an outlet (retention) it is especially important to evaluate the impact.

Regardless of the type of stormwater management facility and measures used on-site, some off-site impacts from urban stormwater use of a wetland are inevitable. By evaluating alternative routes for stormwater, local governments can play an important role in minimizing the impacts.

ROLE OF LOCAL GOVERNMENTS IN WETLAND PROTECTION AND USE

Local governments have a stewardship role in assuring the environmentally wise use of wetlands. Advance studies and planning for wetlands use can provide critical information.

The following activities are recommended:

a. Field surveys should be carried out to identify high quality wetlands which should not be altered.

b. Small watershed studies should be conducted to identify the role of wetlands in the hydrologic regime. Are wetlands strategically located for either flood control purposes or water quality control purposes? If yes, then wetland use presents an opportunity for public benefits.

c. The cumulative effects of wetlands loss and alteration should be considered. A community-wide perspective should be maintained, even while evaluating the impacts of a specific proposal.

d. Basic information should be provided to developers about the importance of wetlands protection and the potential for

wetlands use. It is especially important to put developers "on notice" that there are building constraints before land is purchased. A lot of the controversy may disappear from state and federal regulatory programs when developers no longer over-invest in properties with unbuildable areas such as floodplains and wetlands.

e. The identification of the natural drainage system and some field verification of wetlands locations can be of assistance to developers and the community. Such information is most efficiently prepared for individual small watersheds. Maps which are prepared may be sold to developers to help recover some of the costs.

LOCAL GOVERNMENT INVOLVEMENT WITH STATE OR LOCAL REGULATION

Local governments can be involved in state regulatory actions and/or administration of local wetland ordinances in several ways:

a. Local review of development proposals can identify potential wetlands alterations and evaluate projects with potential erosion impact on wetlands. It can also assure applications for the necessary state permits.

b. If a local government wants to influence state decision-making or individual permits, it is most effective to initiate a local review concurrently with the state-level review. Joint field visits or discussions may be arranged so that the developer is informed of both state and local interests at the same time.

   To the extent possible, state and local government staff should act as partners in the stormwater management and wetlands review process. State agency staff can assist local governments by providing technical reviews and information which may not be available to the local government. In all cases, state agency staff should communicate directly to local officials about the outcome of state wetland permit reviews, and local officials should inform the state about local policies and recommendations concerning state permits. A coordinated state-local government review system will save time and money for public agencies and for private land owners.

c. Local zoning and building inspectors who normally visit a development project at various times can watch for compliance with permit stipulations or failure to maintain wetlands protection and erosion control measures during construction.

d. Local governments may choose to adopt and administer local wetlands ordinances. The Michigan Wetlands Protection Act explicitly provides for local ordinances that are consistent, but perhaps more stringent than state laws. DNR policy is to not issue a state permit when the local wetlands permit has been denied.

Comprehensive Local Management

Stormwater management is the vehicle for integrating municipal master plans for land use and water use. The management of stormwater runoff and erosion control are inseparable as are the management of stormwater and wetlands. Municipal small watershed studies provide the framework for review of site development proposals. A local Stormwater Management Ordinance establishes the requirements for design of on-site stormwater systems compatible with the overall municipal system.

Ordinance Purposes and Performance Standards Related to Wetlands

The planning committees in the three pilot communities participating in the Clinton River Watershed Council project (after review of local land and water resources and existing stormwater problems in the community) listed stormwater management objectives which became the ordinance purposes. Project staff then linked these objectives with performance standards. At this date, enactment of the draft Stormwater and Erosion Control Ordinance is completed in one of the townships, a precedent in Michigan.

Selected ordinance purposes related to wetlands use include:

A. To protect public health, safety and welfare by requiring stormwater management whenever new, expanded or modified developments are proposed.

B. To promote for the most efficient and beneficial uses of land and water resources.

C. To assure that stormwater runoff from development is controlled so that lake and stream water quality is protected, siltation minimized, and flooding problems avoided.

D. To use the natural drainage system for conveying and receiving stormwater runoff, and to minimize the need to construct storm drain pipes.

E. To encourage multiple-purpose stormwater

management which enhances the environmental character of the community.

F. To allow wetlands to be used for stormwater detention in selected locations, while ensuring that the natural functions and quality of wetlands throughout the municipality are protected to the maximum feasible extent.

G. To recognize private responsibility for incorporating stormwater management systems into the early stages of site planning and design.

H. To ensure that all stormwater conveyance and detention facilities will be properly maintained.

Performance standards are included in the ordinance to recognize the variability of circumstances and allow for design flexibility. Specific engineering and design standards are incorporated in a Procedures Manual which may be updated as the technology advances and local planning continues.

The following performance standards have been developed:

Discharge of Stormwater Runoff to Wetlands.

1. Stormwater runoff discharged to wetlands must be diffused to non-erosive velocities before it reaches the wetland.

2. Wetlands may be used for stormwater detention if the following conditions are met:

   a. The wetland storage or detention area is set back at least 100 feet from the edge of any lake or stream.

   b. The wetland does not have significant wildlife habitat or ecological values which would likely be impaired or destroyed.

   c. The wetland has sufficient holding capacity for stormwater, based upon calculations prepared by the proprietor and reviewed and approved by the township engineer.

   d. Adequate on-site soil erosion control is provided to protect the natural functioning of the wetland.

3. During the construction phase of development, adequate erosion control and protection of wetlands shall be provided as required, including such things as:

   a. One or more sediment traps or soil setting basins located in upland locations; and/or

   b. Open-water sediment traps within or adjacent to wetlands, provided that the proposal meets the wetland protection requirements of the Michigan Department of Natural Resources and (municipality) for wetlands alteration.

4. If off site wetlands are used for stormwater management, an easement must be assured in accordance with the requirements of this ordinance.

Other Standards Pertinent to Wetlands.

- Soil erosion control measures shall be installed between the disturbed area and any...wetland.
- Vegetated buffer strips shall be created or retained along the edges of all...wetlands when reasonably determined to be necessary.
- Discharge or runoff from any site which may contain oil, grease, toxic chemicals, or other polluting materials is prohibited unless measures to reduce and trap pollutants meet the requirements of the Michigan Department of Natural Resources and (municipality), based upon professionally accepted principles.

WETLAND UNCERTAINTIES RECOGNIZED IN THIS DEMONSTRATION PROJECT

This demonstration project revealed certain deficiencies in efforts to address local stormwater use of wetlands.

1. A foremost need or problem is the lack effective means to disseminate wetlands information to local level decision-makers. Summary papers are needed that review what is known/what is not known, suggest sound policies for proceeding, and delineate alternatives/associated risks. One good example is "The Use of Wetlands for Stormwater Management and Nonpoint Control: A Review of the Literature", prepared by King County Department of Planning and Community Development for the State of Washington Department of Ecology, October 1986.

2. Lack of wetlands information is applicable in this region. Summaries by experts should be undertaken that indicate generalities and wetlands research results in a specific region.

3. More guidance is needed as to whether "excess" stormwater runoff will significantly alter water levels and response of vegetation to inundation. Simple methods

of assessing specific wetlands sensitivity to water supply fluctuations should be developed.

4. More guidance is needed to help estimate long term impacts and maintenance needs in wetlands used for stormwater management. Guidance on monitoring, maintenance methods, and cost estimates on which to base maintenance agreements/ funds and administrative programs are needed.

5. More guidance is needed to determine what modifications of natural wetlands are acceptable. Practical evaluation methods and design principles are needed.

6. More guidance is needed with regard to the principles for deciding on optimum size and location of wetlands for systematic watershed stormwater management to meet multiple objectives. Are rules-of the-thumb such as maximum watershed size 500 acres/minimum wetland size 5 acres reasonable places to start short of a specific watershed study?

7. More research and guidance is needed to help determine what wetland buffer size is warranted. Simple methods should be developed to correlate wetlands types versus development proposals to decide appropriate buffer size and/or extent of transition zone to be preserved.

SUMMARY

This demonstration project has identified the key aspects of wetlands management: multiple-objective uses of wetlands; the watershed context for wetlands decisions; the role of wetlands in the hydrological system; the need for inter-jurisdictional coordination; involvement and training of key interests and local decision-makers; and building a constituency to support stormwater management. An examination of the full range of local government roles available in the use and protection of wetlands reveals improved opportunities for better wetlands management beyond those offered by wetland regulatory programs.

NOTES

Lillian Dean is the principal author of "Stormwater Management Guidebook for Michigan Communities" from which materials are presented.

chapter eight

Wetlands and Groundwater

Assessing the Relationship of Groundwater and Wetlands

Garrett G. Hollands
IEP Inc.

INTRODUCTION

Wetlands have been assigned a variety of hydrologic functions by various federal, state and local wetland statutes. The Massachusetts Wetlands Protection Act (MGL Chapter 131, section 40) recognizes the groundwater supply function of wetlands and the Wisconsin Administrative Code 1.95 lists five watershed functions including "groundwater". Groundwater functions under Wisconsin NR 1.95 are described as follows:

> "Groundwater may discharge to a wetland, recharge from a wetland to another area, evaporate from and/or flow through a wetland."

The Massachusetts Wetland Protection Act regulations (310 CMR 10.00) state:

> "Groundwater supply means water below the earth's surface in the zone of saturation."

In Massachusetts, the groundwater function of a wetland has traditionally been considered the recharge value of the wetland to the underlying aquifer and the role that wetland soils may serve in preventing polluted surface water from entering the aquifer.

While other slight variations occur nationally in wetland statutes, these two definitions collectively represent the most common concepts of wetland groundwater functions and raise the most common questions with regard to the assessment of wetlands and groundwater relationships. Regulators of wetlands are commonly asked to answer the following questions relating to wetland groundwater functions:

1. Is the wetland discharging water from an aquifer to surface waterbodies or is it recharging water from surface waterbodies to the underlying aquifer?

2. Is the recharge from the wetland to the aquifer important to other wetlands?

3. What role does groundwater play in the water balance of the wetland?

4. What role does the wetland play in supplying water to underlying and adjacent aquifers used for public or private water supplies?

5. What effect does the groundwater function of a wetland have upon other wetland functions, such as flood control, wildlife, etc.?

Hydrology is the primary and critical force that creates and modifies wetlands. An understanding of wetland hydrology is basic to understanding any wetland function (Gosselink and Turner 1978; U.S. Fish and Wildlife Service 1984). Much of the groundwater research in recent years has been generated by the need to solve complex hazardous waste and water supply issues. This has led to a questioning of past theories and the generation of new theories concerning groundwater hydrology. Gathering actual wetland groundwater data is time consuming and expensive, extrapolating data from one wetland to another can be problematic. No quick, accurate and inexpensive groundwater function predictors are available. Even hydrogeologists, experienced in wetland hydrology, cannot consistently predict the hydrogeologic functions of specific wetlands.

DATA REQUIREMENTS

To understand the groundwater function of a specific wetland, a three dimensional understanding of the hydrogeologic framework of the wetland is required. This requires at a minimum, an understanding of the following:

1. Geologic history, including an understanding of the current theories relative to the geologic processes that created the topographic and hydrogeologic setting in which the wetland is located, including bedrock and surficial geology.

2. Stratigraphy of the geologic units underlying the wetland and their physical properties, such as permeability.

3. History, stratigraphy and physical properties of wetland organic or mineral soils (pedological definition of "soil").

4. Description of the wetland vegetative community.

5. Groundwater and surface water hydrology, including a water budget for the wetland based on numbers 1-4 above.

GROUNDWATER RECHARGE AND DISCHARGE

Wetlands are generally considered by hydrologists to be discharge areas in terms of total water budget. However, when only the groundwater component of the water budget is investigated, some wetlands function primarily as recharge areas, some as discharge areas, and some vary from recharge to discharge areas seasonally. Recharge and discharge may be occurring at the same time in some wetlands. The recharge/discharge relationship of a wetland is a function of groundwater piezometric surface ("head") relationships and antecedent conditions.

To determine head relationships, nested water table observation wells (piezometers) are required, which permit simultaneous measurements of head at various levels within the aquifer. Measurements for at least one water year may be required to establish a complete record of recharge/discharge functions of a wetland. Costs are normally high. (Definitions of recharge and discharge are contained in Freeze and Cherry, 1979).

Many wetlands have regional discharge and recharge relationships. Water recharging an aquifer for one wetland may, in part, discharge to a downgradient wetland. To determine the hydrologic support function of a wetland, the wetland must be placed in a regional hydrogeologic context (Winter 1976). This requires a large amount of costly hydrogeologic data not normally available to the wetland regulator. Concord, Massachusetts is an exception. The Town has detailed 1" = 800 ft. scale hydrogeologic maps (IEP, 1978) which portray the three dimensional aspects of Concord's aquifers. This data is based on thousands of subsurface data points including borings and seismic profiles. Based upon such data, the hydrogeologic relationships of wetlands (Hollands and Mulica 1978) can be predicted with a high degree of scientific certainty. Without a similar amount of hydrogeologic data, or without nested wells located in the wetland, prediction of wetland recharge/discharge relationships is, at best, a guess.

Perched wetlands, water table wetlands and other hydrogeologic classifications such as artesian and water table/artesian wetlands (Motts and O'Brien 1980) also require nested wells for identification. Hydrogeologic classification of wetlands is important in understanding a wetland's water balance and the effect of hydrology on other wetland functions. The wetland hydrogeologic classification that appears to be most used by non-hydrogeologist wetland regulators is that of Novitzki (1978).

Novitzki classified wetlands in Wisconsin as "surface water depression", "ground water depression", "surface water slope" or "ground water slope". This classification combines topography, surface water, and ground water parameters. However, without wetland-specific hydrogeologic data, it is doubtful if this method can be accurately applied by non-hydrogeologists.

WETLANDS SOILS AND GROUNDWATER

Wetland soils that are predominantly organic-rich (sapric, fibric and hemic soils) have low vertical permeabilities, commonly lower than the underlying aquifer. This permeability difference may decrease water discharge rates from discharge wetlands and may create water table/artesian conditions (Motts and O'Brien 1980). In recharge wetlands, the low permeability, high porosity wetland soils may regulate leakage (recharge) to the aquifer. The role of organic soils to influence recharge/discharge functions relationships needs to be investigated further.

SUMMARY

There are no short cuts to accurately predict the groundwater function of a wetland. Detailed hydrogeologic data is necessary. The data is expensive and may require a year or more of groundwater elevation observations before recharge/discharge relationships can be understood. The state-of-the-art of understanding wetland water budgets is in its beginning stages and much more research is needed before accurate predictors are developed for non-hydrogeologists. Since hydrology is the driving force of wetland functions and reliable wetland hydrology predictors do not exist, it is doubtful if accurate assessments can be made of many other wetland functions. There are few short cuts to understanding wetland hydrologic functions. Detailed, wetland-specific, multi-disciplinary investigations conducted by qualified scientists are needed.

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Inundation Characteristics of Wet Prairie and Marsh Wetlands in Southwestern Florida

Brian H. Winchester
Winchester Environmental Associates, Inc.

James S. Bays and John C. Higman
CH2M HILL, Inc.

INTRODUCTION

Florida contains a diverse array of wetlands that have often been the focus of regulatory attention and scientific study. However, most of the wetland studies to date have placed a greater emphasis upon describing biological characteristics than upon defining hydrologic regimes. Consequently, the inundation characteristics of Florida wetlands are still poorly understood. Understanding wetland inundation characteristics in Florida is further complicated by the vast diversity of wetlands in the state and the fact that many wetland systems have been hydrologically altered. This study of the marsh and wet prairie wetlands of southwestern Florida provides a preliminary hydrologic understanding of an important wetland group and also suggests a study approach which may be applicable to other wetlands in the state.

DESCRIPTION OF STUDY AREA

All study wetlands were located on the Ringling MacArthur Reserve, a 33,000-acre tract of land in southwestern peninsular Florida (see Figure 1). The Reserve contains over 1100 individual wetlands, ranging in size from a quarter-acre to over 100 acres. Virtually all of the wetlands are freshwater marsh or wet prairie systems. Shrub swamp and wooded swamp systems are present but uncommon. Many of the Reserve wetlands are hydrologically isolated, receiving inflow through rainfall, surface runoff and lateral groundwater flow. Other Reserve wetlands are part of large regional drainageways (known locally as sloughs) which serve as major conduits of water during the wet season. The southeastern section of Deer Prairie Slough, the largest slough on the Reserve, is partially drained by agricultural ditches and supports a flora indicative of hydrologic alteration (Winchester unpublished data).

The physiography of Reserve wetlands is described in Winchester et al. (1985). The depth of the wetlands examined ranged from 0.6 to 2.5 feet below the wetland-upland edge, with an average maximum depth of 1.7 feet below the wetland-upland edge. Surficial soils were typically sandy around the wetland perimeter and at wetland depths of less than one foot. Organic soils predominate in the deeper wetland interiors.

Wetlands on the Reserve exhibit distinct vegetation zonation patterns. The common vegetation zones and the dominant plant species within each were identified in Winchester et al (1985) as: the Hypericum zone (Hypericum fasciculatum); the Panicum-Rhynchospora zone (Panicum hemitomon, Panicum rigidulum, and Rhynchospora tracyi); the Mixed Emergent zone (Panicum hemitomon, Pontederia cordata, and Sagittaria lancifolia); the Cladium zone (Cladium jamaicensis); the Cephalanthus zone (Cephalanthus occidentalis); and the Fraxinus-Salix zone (Fraxinus caroliniana and Salix caroliniana).

METHODS

Water levels were measured twice-monthly at 26 wetlands on the Reserve from April 1985 to September 1986. A shallow well was installed in each wetland at the boundary between upland and wetland vegetation (i.e., the wetland rim). Another shallow well was located in the interior of the wetland. Well holes were hand-augered to a depth of 7 - 8 feet in rim wells and 4 - 5 feet in interior wells. Where possible, interior wells were extended into mineral soils beneath surficial peat layers. Wells consisted of 1.25-inch PVC pipe with 2 foot sections of 0.010-inch slotted screen at the bottom. Relative well site and upland edge elevations were determined using standard survey techniques. Staff gages were installed adjacent to the interior wells in fifteen of the wetlands to verify that within-casing water levels were comparable to actual water levels.

Where staff gages were present, they were used to determine aboveground wetland water levels. In wetlands lacking staff gages, aboveground water levels were determined from reading interior wells. Although interior well readings did not identically track staff gage

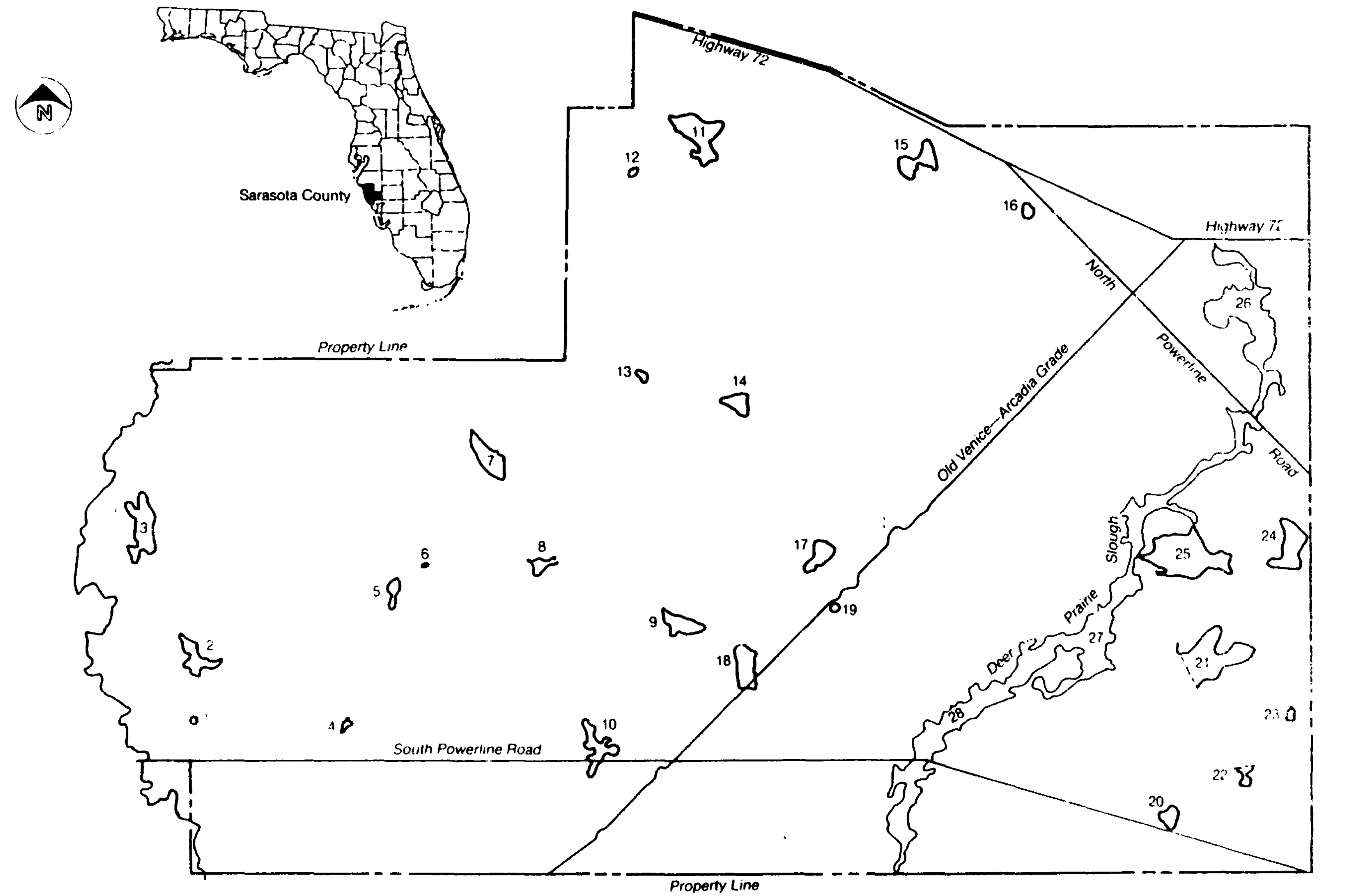
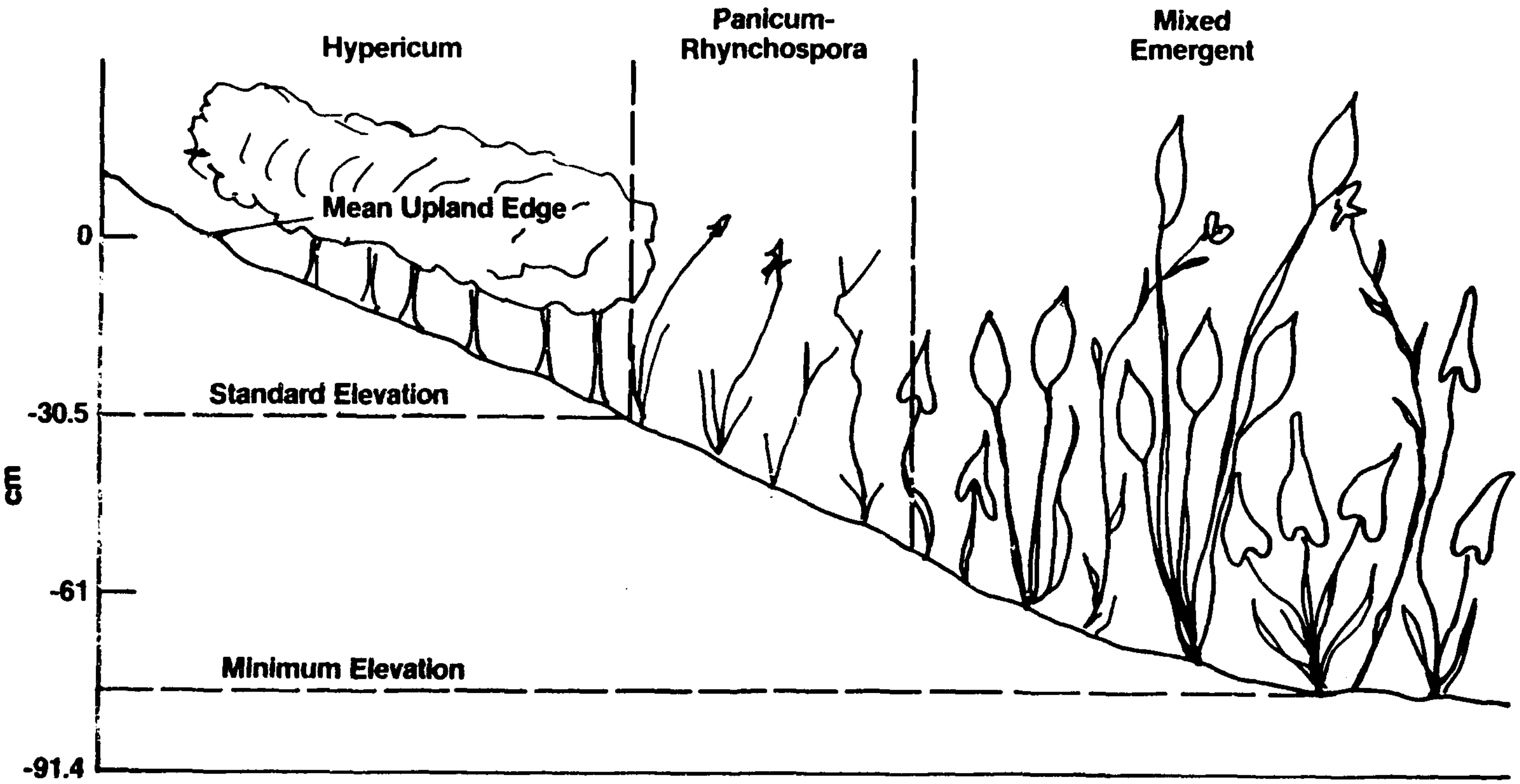


FIGURE 1.
Location of Study Wetland on the Ringling MacArthur Reserve.



Note: X-axis not to scale.

FIGURE 2.
Generalized Wetland Elevation Parameters.

readings, they were typically within 0.1 to 0.2 feet of staff gage readings for most study wetlands. Appreciable differences between staff gage and interior well water levels ocurred only in study wetlands 1 to 10. Water levels less than minimum staff gage elevations were always determined from interior well readings.

For this analysis, water level data were examined on a water year (WY) basis, which runs from October 1 to September 30 of the following year. Because the monitoring program did not commence until April 1985, complete data were not available for WY 1985. However, Reserve wetlands were qualitatively observed throughout WY 1985, and maximum and minimum wetland water levels for that water year both occurred during the April-September monitoring period. Consequently, maximum and minimum wetland water levels were determined from monitoring data for both WY 1985 and WY 1986.

Frequency distributions indicated that much of the hydrologic data from RMR wetlands did not show a normal distribution. Consequently, a non-parametric approach was used for statistical comparisons, thereby avoiding the need to verify normality or transform data. The major tests utilized were the Wilcoxon Rank Sum Test (WRST) for independent groups, and the Signed Rank Test (SRT) for paired data (Gertz, 1978; Mendenhall and Sincich, 1984). All statistical calculations were made with the EPISTAT PC program (Gustafson, 1984). Significance levels were set at <0.05, though test statistics less than 0.01 were indicated where relevant.

The hydrologic characteristics of the study wetlands were examined relative to a number of significant elevations. The mean upland edge elevation was an average elevation taken from the endpoints of wetland topographic transects, as defined by the junction of the wetland with a palmetto rim, a pine forest, or an oak-palm hammock. It is typically close to and sometimes identical with the ground elevation of the rim well. The minimum elevation was the lowest elevation measured in the wetland. Although topographic transects were laid out to cross the deepest zones of each study wetland, the measured minimum elevation probably did not always represent the single lowest point in the wetland, especially in very large wetlands. Because of this and because of the topographic variation between wetlands, wetland hydrologic characteristics were also examined relative to a standard point 1.0 foot below the mean upland edge of each wetland. This was termed the standard elevation. The use of the standard elevation as a reference point allowed the hydrologic comparison of different wetlands on the Reserve without the confounding factor of variation in landscape elevation, wetland topography, and basin depth. An idealized wetland profile showing these elevational points is provided in Figure 2.

RESULTS AND DISCUSSION

The analysis of the Reserve study wetlands focused upon five primary hydrologic features. These were: 1) water level hydrographs, 2) average water depth, 3) maximum flooding depth, 4) maximum drydown depth, and 5) hydroperiod or duration of flooding. Because study wetlands 18, 20, 22, 27 and 28 showed clear evidence of drainage-related hydrologic alteration, they were excluded from most of the present analysis. Hydrologic data were not collected from study wetlands 13 and 23.

Water Level Hydrographs

In order to evaluate overall patterns of water level fluctuation in the study wetlands, individual wetland hydrographs were plotted relative to standard elevations (Figure 3). The hydrographs show that much greater water level variation occurred between wetlands during the dry season than during the wet season. The severe regional drought which preceded the monitoring period continued through June, 1985. Water levels dropped substantially belowground during this time, with a 3 to 5-foot spread in water levels generally encountered on any given sampling episode. With the onset of the 1985 wet season, wetland water levels rose sharply, emerged aboveground, and thereafter exhibited much less inter-wetland variation. During the 1985 wet season, wetland water levels were generally within a 1-foot range on any given sampling episode. The 1986 dry season and wet season repeated these patterns, though the 1986 dry season was shorter than that of 1985.

Considering the amount of physiographic variation between study wetlands, it is interesting to note the similarity of individual wetland hydrographs during the wet season. If individual wetland features such as size, basin morphology, hydrologic contiguity, and relative location in the drainage basin were primary factors controlling wet season water levels, more variation in wetland hydrographs would be expected. Instead, the wetlands show parallel responses to regional climatic conditions, despite individual site differences.

Other data from the Reserve indicate a close relationship between wet season wetland water levels and the elevation of the surficial water table in adjoining uplands (Winchester et al, 1989a). The relationship probably also contributes to the observed similarity of wetland hydrographs. With the patchy distribution of summer rainfall on the Reserve (Dames & Moore, 1986), and the study wetlands being widely scattered over the 33,000-acre tract, considerable between-wetland hydrograph variation would be

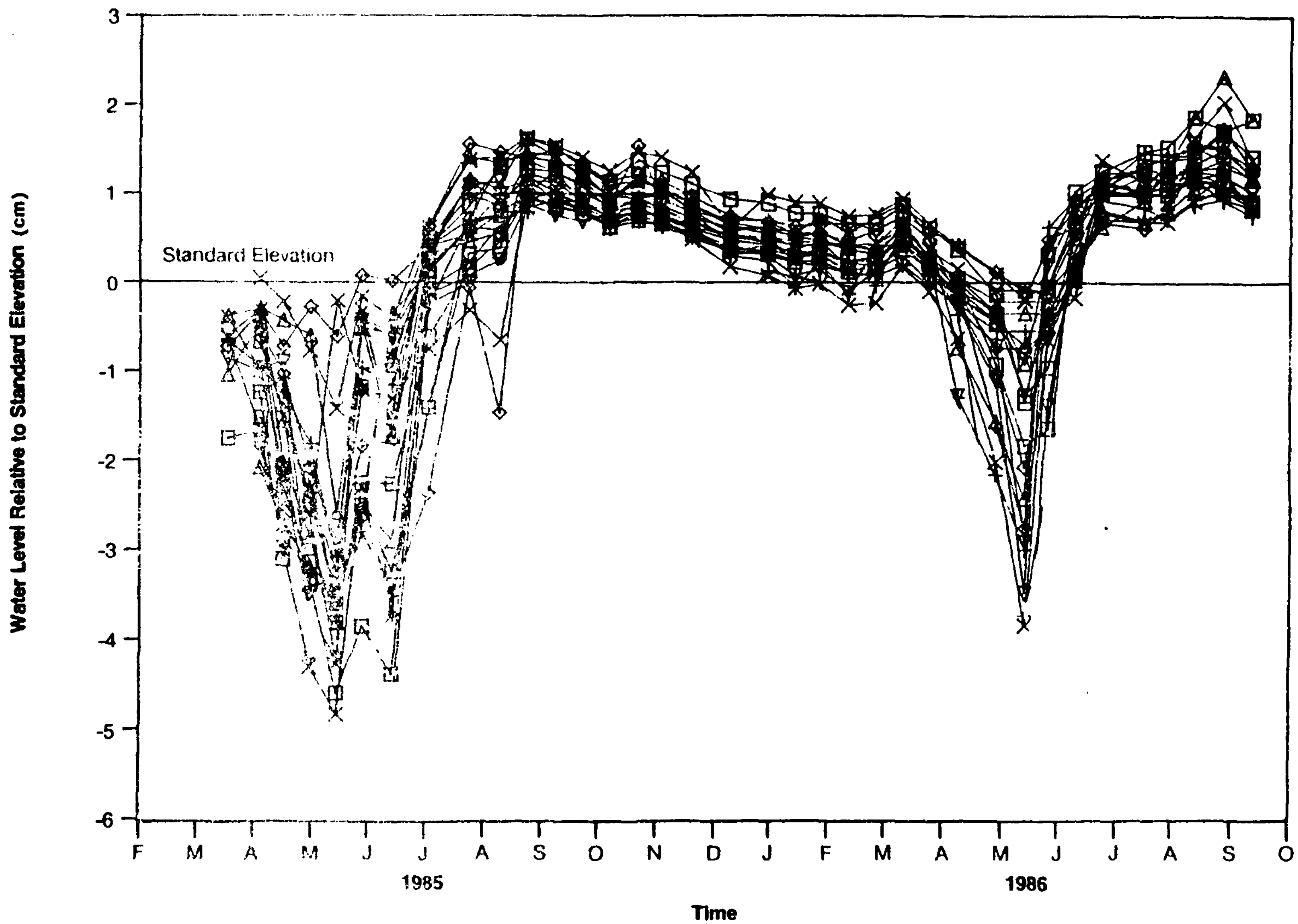


FIGURE 3.
Hydrographs of 21 Study Wetlands.

expected if wetland recharge occurred only by rainfall and direct runoff. Conversely, less hydrograph variation would be expected if equalizing groundwater flows occurred between the wetland and the adjoining upland water table.

Individual site characteristics apparently became more important in determining water levels during the dry season. as water levels drop to or below the wetland surface. The greater variation in individual wetland hydrographs during the dry season is at least partly due to differences in wetland storage, which is a function both of basin depth and soil porosity. Because many of the study wetlands exhibit dry season water level mounds relative to the upland water table (Winchester et al, 1989a), lateral and vertical soil conductivities and the depth of interior organic soils are also thought to be important. Finally, there may also be differences in the amount of evapotranspiration occurring in different wetlands, though this becomes less important as water levels drop further below the wetland surface.

Average Water Depth

Average water depth (AWD) provides an integration of a wetland's duration of flooding (or hydroperiod) and average flooding depth. It is calculated by taking the numerical average of all above- and below-ground water level measurements. relative to standard elevation, for a 12-month period. An annual "combined" average water depth has proven to be a very useful parameter for statistically distinguishing between hydrologically altered and unaltered wetlands (Winchester et al, 1989b), and may also be useful as a general descriptor of wetland hydrology.

The average water depths of study wetlands WY 1986 are provided in Table 1. Average water depths in the 21 study wetlands ranged from 0.20 and 0.88 feet during WY 1986. The combined AWD of these 21 wetlands during WY 1986 was 0.54 feet.

Hydroperiod

Hydroperiod is defined here as the duration (in days per year) of flooding at a given wetland elevation. Water levels at or above ground surface constitute flooding. Conversely, drydown period is defined as the number of days per year during which water levels are below the wetland surface.

Hydroperiod data for the 21 study wetlands during WY 1986 are presented in Table 2.

Table 1
AVERAGE WATER DEPTHS OF STUDY WETLANDS DURING WY 1986

Study Wetland	Average Water Depth (in feet) Relative to Standard Elevation[1]	Average Water Depth (in feet) Relative to Minimum Elevation[2]
1	0.61	2.11
2	0.29	0.99
3	0.70	1.40
4	0.71	0.21
5	0.86	1.86
6	0.45	1.15
7	0.70	1.05
8	0.36	0.96
9	0.45	1.00
10	0.58	1.38
11	0.41	0.81
13	0.35	1.25
14	0.42	0.92
15	0.31	1.01
16	0.61	1.41
17	0.45	1.05
19	0.20	1.00
21	0.88	0.78
24	0.82	1.02
25	0.51	0.81
26	0.57	2.87

[1]Standard elevation is defined as a point 1.0 foot below the upland-wetland edge.

[2]Minimum elevation is the lowest elevational point encountered in topographic transects across study wetlands.

Table 2
HYDROPERIOD AND DRYDOWN PERIOD OF STUDY WETLANDS

Study Wetland	WY 1986 Hydroperiod Relative to Standard Elevation Days	WY 1986 Hydroperiod Relative to Minimum Zone Days	WY 1986 Drydown Period Relative to Standard Elevation Days	WY 1986 Drydown Period Relative to Minimum Zone Days	Minimum Estimate of 1985 Drydown Period Relative to Standard Elevation Days	Minimum Estimate of 1985 Drydown Period Relative to Minimum Zone Days
1	319	365	46	0	122	76
2	289	319	76	46	106	106
3	335	335	30	30	122	122
4	319	319	46	46	91	91
5	350	365	15	0	91	15
6	304	335	61	30	91	61
7	319	350	46	15	91	91
8	304	350	61	15	91	91
8	304	335	61	30	91	91
10	319	350	46	15	91	76
11	319	350	46	15	122	76
13	304	350	61	15	122	61
14	319	335	46	30	122	122
15	304	350	61	15	106	61
16	304	350	61	15	76	0
17	319	350	46	15	106	61
19	243	335	122	30	106	61
21	321	321	44	44	91	91
24	350	365	15	0	91	91
25	319	319	46	46	106	106
26	304	365	61	0	152	30

Hydroperiod estimates could not be generated for WY 1985 because water level monitoring did not begin until the middle of the water year. However, minimum estimates of the 1985 drydown period have been generated, since the monitoring encompassed much of the 1985 dry season and allows a general comparison between the two years. For many wetlands the minimum 1985 drydown period underestimates the actual drydown period by an unknown amount, but the data are nevertheless of value in establishing natural ranges of wetland drydown periods. Because of the semi-monthly nature of the water level monitoring program, the precision of hydroperiod estimates is plus or minus one week.

The average WY 1986 hydroperiod of the 21 study wetlands, relative to standard elevation, was 313 days. The range in hydroperiod for individual wetlands was 289 to 350 days, with over 75% of the study wetlands falling within the range of 304 to 321 days. During WY 1986, the corresponding wetland drydown period averaged 55 days. Relative to minimum elevation, the WY 1986 hydroperiod and drydown period averaged 341 and 24 days, respectively.

For WY 1985, the drydown period relative to standard elevation was at least 104 days and ranged from 76 to 152 days for individual wetlands. This estimate of drydown period should be considered a minimum value because water levels were observed to drop belowground in some study wetlands shortly before monitoring began (i.e. April 1985). When compared against the average WY 1986 drydown period of 55 days, the effect of the 1984-85 drought on wetland hydroperiod is apparent.

Maximum Flooding Depths

Maximum flooding depths of study wetlands are presented in Table 3. Because the monitoring period encompassed the highest and lowest water levels occurring during both WY 1985 and WY 1986, maxima and minima are presented for both years. The average maximum water depth of the five hydrologically altered wetlands (i.e study wetlands 18, 20, 22, 27, and 28) was not significantly different ($p>0.05$, WRST) from that of the 21 unaltered wetlands in either 1985 or 1986. Consequently, data from all of the study wetlands were used in the analysis of maximum flooding depths.

In WY 1985, maximum flooding depths relative to standard elevation averaged 1.23 feet for all wetlands combined and ranged from 0.34 to 1.60 feet. In 1986, maximum flooding depths averaged 1.52 feet at standard elevation and ranged from 1.00 to 2.34 feet. This difference was highly significant ($p<0.01$, SRT), indicating that Reserve wetlands had higher maximum water levels in 1986 than in 1985.

A closer examination of the data in Table 2 reveals that there was little difference in maximum flooding depths in the wetlands west of Deer Prairie Slough (i.e., study wetlands 1 through 19). Taken together, the maximum flooding depth of these wetlands averaged 1.28 feet in 1985 versus 1.35 feet in 1986, a 0.07 foot difference that was not significant ($p>0.05$, SRT). For these study wetlands, the greatest individual increase in maximum flooding depth from 1985 to 1986 was 0.31 feet (at study wetland 16), and a number of the wetlands showed maxima in 1986 which were lower than in 1985.

For the wetlands east of and along Deer Prairie Slough (i.e. study wetlands 20 through 28), maximum flooding depths relative to standard elevation averaged 1.13 feet in 1985 versus 1.90 feet in 1986, a highly significant difference of 0.77

Table 3
MAXIMUM FLOODING OF STUDY WETLANDS DURING WY 1986[1]

Study Wetland	Maximum Flooding Depth Relative to Standard Elevation[2] 1985	1986	Maximum Flooding Depth Relative to Minimum Zone 1985	1986
1	1.28	1.30	2.78	2.80
2	1.04	1.19	1.74	1.89
3	1.43	1.46	2.13	2.16
4	1.32	1.47	0.82	0.97
5	1.59	1.42	2.59	2.42
6	1.60	1.59	2.30	2.29
7	1.42	1.68	1.77	2.03
8	1.25	1.00	1.85	1.60
9	1.46	1.45	2.01	2.00
10	1.48	1.51	2.28	2.31
11	0.92	1.06	1.32	1.46
13	0.89	1.27	1.79	2.17
14	0.95	1.17	1.45	1.67
15	1.06	1.24	1.76	1.94
16	1.43	1.74	2.23	2.54
17	1.33	1.17	1.93	1.77
18	1.17	1.21	2.17	2.21
19	1.37	1.39	2.17	2.19
20	0.34	1.22	1.64	2.52
21	1.38	2.11	1.28	2.01
22	0.80	1.55	1.10	1.85
24	1.16	1.72	1.36	1.92
25	1.41	2.34	1.71	2.64
26	1.12	2.15	3.42	4.45
27	1.38	2.11	2.48	3.21
28	1.42	1.98	3.42	3.98

[1]All depths in feet.
[2]Standard elevation is defined as a point 1.0 foot below the wetland-upland edge.

Table 4
MAXIMUM DRYDOWN DEPTHS OF STUDY WETLANDS DURING WY 1986[1]

Study Wetland	Maximum Drydown Depth Relative to Standard Elevation[2] 1985	1986	Maximum Drydown Depth Relative to Minimum Zone 1985	1986
1	-3.58	-0.91	-2.08	+0.59
2	-3.80	-2.99	-3.10	-2.29
3	-3.38	-2.55	-2.68	-1.85
4	-3.31	-2.27	-3.81	-2.77
5	-1.47	-0.30	-0.47	0.70
6	-4.22	-3.47	-3.52	-2.77
7	-3.58	-1.54	-3.23	-1.19
8	-2.85	-1.20	-2.25	-0.60
9	-3.34	-2.16	-2.79	-1.61
10	-3.80	-1.51	-3.00	-0.71
11	-4.26	-2.48	-3.86	-2.08
13	-4.21	-2.90	-3.31	-2.00
14	-3.81	-1.26	-3.31	-0.76
15	-3.09	-1.44	-2.39	-0.74
16	-0.99	-1.08	+0.17	-0.28
17	-2.78	-0.86	-2.18	-0.26
19	-4.83	-3.83	-4.03	-3.03
21	-4.19	-1.42	-4.29	-1.52
24	-3.02	-0.22	-2.82	-0.02
25	-3.59	-2.36	-3.29	-2.06
26	-2.48	-2.22	-0.18	+0.08

[1]All depths in feet.
[2]Standard elevation is defined as a point 1.0 foot below the wetland-upland edge.

feet ($p<0.01$, SRT). The smallest increase from 1985 to 1986 in these flooding depths was 0.56 feet (study wetlands 24 and 28); the largest was 1.03 feet (study wetland 26). Part of this difference can be attributed to the low 1985 maxima exhibited by study wetlands 20 and 22 (0.34 and 0.80 feet, respectively). However, the 1985 maxima of the other eastern study wetlands were comparable to the maxima of wetlands west of Deer Prairie Slough. The remainder of the 1985-1986 differences for these eastern wetlands can be attributed to the very high 1986 maximum flooding depths encountered at Deer Prairie Slough and at the other large study wetlands to the east.

If maximum flooding depths are examined relative to the minimum elevation of each wetland, maxima averaged 1.98 feet in 1985 and ranged from 0.82 feet to 3.42 feet. In 1986, maximum flooding depths relative to minimum elevation averaged 2.27 feet, ranging from 0.97 feet to 4.45 feet.

Maximum Drydown Depths

Maximum drydown depths relative to standard elevation averaged -3.36 and -1.86 feet for the 21 unaltered study wetlands in 1985 and 1986, respectively (Table 4). This difference was significant ($p<0.05$, SRT) indicating that substantially greater drydown depths occur during drought years. Maximum drydown depths ranged from -0.73 to -4.83 feet in 1985, and from -0.22 to -3.83 feet in 1986.

Maximum drydown depths relative to minimum elevation averaged -2.69 and -1.20 for 1985 and 1986, respectively. All of the study wetlands dried down in 1985 except for study wetland 16, and incidental observations throughout the Reserve failed to identify any other natural wetlands which retained aboveground water during the peak of the drought (i.e. late May and early June). At this time the only surface water observable on the Reserve was in some of the deeper roadside ditches and excavated watering pits. In study wetland 16, water levels dropped to within 0.1 foot of the minimum elevation, but did not drop belowground. The maintenance of higher water levels at this wetland may have been related to the exceptionally dense thatch of pickerelweed (Pontederia cordata) which covered the wetland surface. This thatch, left over from the 1984 growing season, was very thick and probably precluded significant evaporation. Significant regrowth of living wetland vegetation did not occur until after the drought ended, so transpiration from the wetland was probably minimal. Except for study wetland 26 (which had the deepest minimum elevation of all the study wetlands--3.30 feet below the mean upland edge) and study wetland 5, 1985 maximum drydowns exceeded 2.0 feet below minimum elevation in all other study wetlands. In 1986, maximum drydown depths relative to minimum elevation were more variable, with three of the study wetlands (i.e. study wetlands 1, 5, and 26) maintaining surface water throughout the water year.

SUMMARY AND CONCLUSIONS

Hydrologic monitoring was conducted on 26 study wetlands in southwestern Florida from April 1985 to September 1986. The subsequent analysis of monitoring data focused upon five primary hydrologic features: 1) water level hydrographs, 2) hydroperiod or duration of flooding, 3) average water depth, 4) maximum flooding depth, and 5) maximum drydown depth.

Even though the study wetlands varied considerably in size, basin physiography, and hydrologic contiguity, they still exhibited similar patterns of water level fluctuation. During the wet season, wetland water levels varied primarily in response to regional climatic conditions, with little variation in relative water levels from one wetland to another. During the dry season, greater variation in water levels occurred between wetlands, and the importance of individual site characteristics in determining wetland water levels increased.

Average water depths of the study wetlands at standard elevation (i.e., one foot below the upland-wetland edge) ranged between 0.20 and 0.88 feet during water year 1986. The overall average of these values for all study wetlands combined was 0.54 feet. The average hydroperiod of the study wetlands in water year 1986, relative to standard elevation, was 313 days. The range in hydroperiod for individual wetlands was 289 to 350 days, with over 75% of the study wetlands falling within the range of 304 to 321 days.

In water year 1985 (a drought year), maximum flooding depths relative to standard elevation averaged 1.23 feet for all wetlands combined and ranged from 0.34 to 1.60 feet. In 1986 maximum flooding depths were significantly higher, averaging 1.52 feet at standard elevation and ranging from 1.00 to 2.34 feet for individual wetlands. Maximum drydown depths relative to standard elevation averaged -3.36 and -1.86 feet for the study wetlands in 1985 and 1986, respectively. This difference was significant, indicating that substantially greater drydown depths occur in wetlands during drought years. Maximum drydown depths ranged from -0.73 feet to -4.83 feet in 1985, and from -0.22 to -3.83 feet in 1986.

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Gcomorphological Controls on Subsurface Transport in Two Salt Marshes

Willia.n K. Nuttle and Judson W. Harvey
Department of Environmental Sciences
University of Virginia

INTRODUCTION

Salt marshes are among the most productive ecosystems on earth, primarily because of the high productivity of the salt marsh grasses. As environmental scientists, we are interested in the factors that control the rate of production of the grasses and the pathways by which the energy fixed in the plant biomass is distributed throughout the marshes and exported to adjacent estuaries. As stewards of the environment, we are interested in the benefits realized from this high productivity and the measures that best protect marshes from disturbance as well as techniques to return disturbed areas to their natural state. As engineers, we marvel at Nature's ingenuity and wonder how we can duplicate it and use it to our advantage, for instance in wastewater treatment.

Whatever our perspective, we must be aware of the critical role played by the subsurface hydrology in coastal marshes. Water movement through the sediment controls the availability of nutrients, controls the build up of toxins, such as sulfides, and controls the degree of aeration of the sediment, which has recently been shown to be a critical factor in determining the rate of above-ground productivity (Chalmers, 1982; Howes et al, 1986). We can expect that factors controlling water movement in the sediment will also affect the productivity of the vegetation, and these factors should be important in future investigations and descriptions of coastal marshes. We have undertaken investigations in two vastly different salt marshes for the purpose of identifying the common factors important to subsurface hydrology.

THE WATER BUDGET

The elements of the water budget of the sediment in a salt marsh are shown schematically in Figure 1. Salt marsh sediments receive water from direct precipitation on the marsh, runoff from adjacent upland areas, and possibly through groundwater discharging from regional aquifers. Tides in adjacent estuaries cause periodic flooding of the marsh surface and infiltration of the tidal water into the sediment, and tidally driven fluctuations in head may cause a periodic exchange of groundwater with an underlying aquifer. Water is lost from the sediments by evapotranspiration and by lateral drainage through the sediment and out into the tidal creeks. Tidal fluctuations in creek level also affect the direction and magnitude of water fluxes across the creekbanks.

In each of the two marshes we have studied, the sediment is isolated from interaction with an underlying aquifer by a relatively impermeable layer of material beneath the marsh sediments. We focus our attention on describing the extent of pore water drainage from the sediment into the creeks and on the relative importance of drainage and evapotranspiration as mechanisms for water loss from the sediment. This affects the turn-over rates of water in the sediment and the degree to which the chemistry of the creek water affects the biogeochemistry of the sediment. The method used in our studies was to observe the dynamic behavior of hydraulic head (related to pore water pressure), which relates to changes in water content and the rate of water movement in the sediment. We then used these data to test models of the water balance in the sediment. We infer the critical factors affecting subsurface hydrology from the models that best explain the observed dynamics of hydraulic head.

FIELD STUDIES

Belle Isle Marsh

Belle Isle marsh (N42°22', W71°02') is the last remnant (100 hectares) of the once extensive marshes in Boston, Massachusetts. The sediments consist of a layer of clayey peat (10% ash-free dry weight, Chen 1986) 70 cm deep which grades into a layer of gray, silty clay. A layer of impermeable clay underlies the study area at 170 cm. The vegetation is a mix of Distichlis spicata, Spartina patens and Spartina alterniflora. The tall growth form of Spartina alterniflora is found along the creek banks. The hydrologic regime is controlled by periods of daily flooding of the marsh surface during spring tides alternating with neap-tide periods in which the surface does not flood for several days. The average tidal range in the creeks is 2 m.

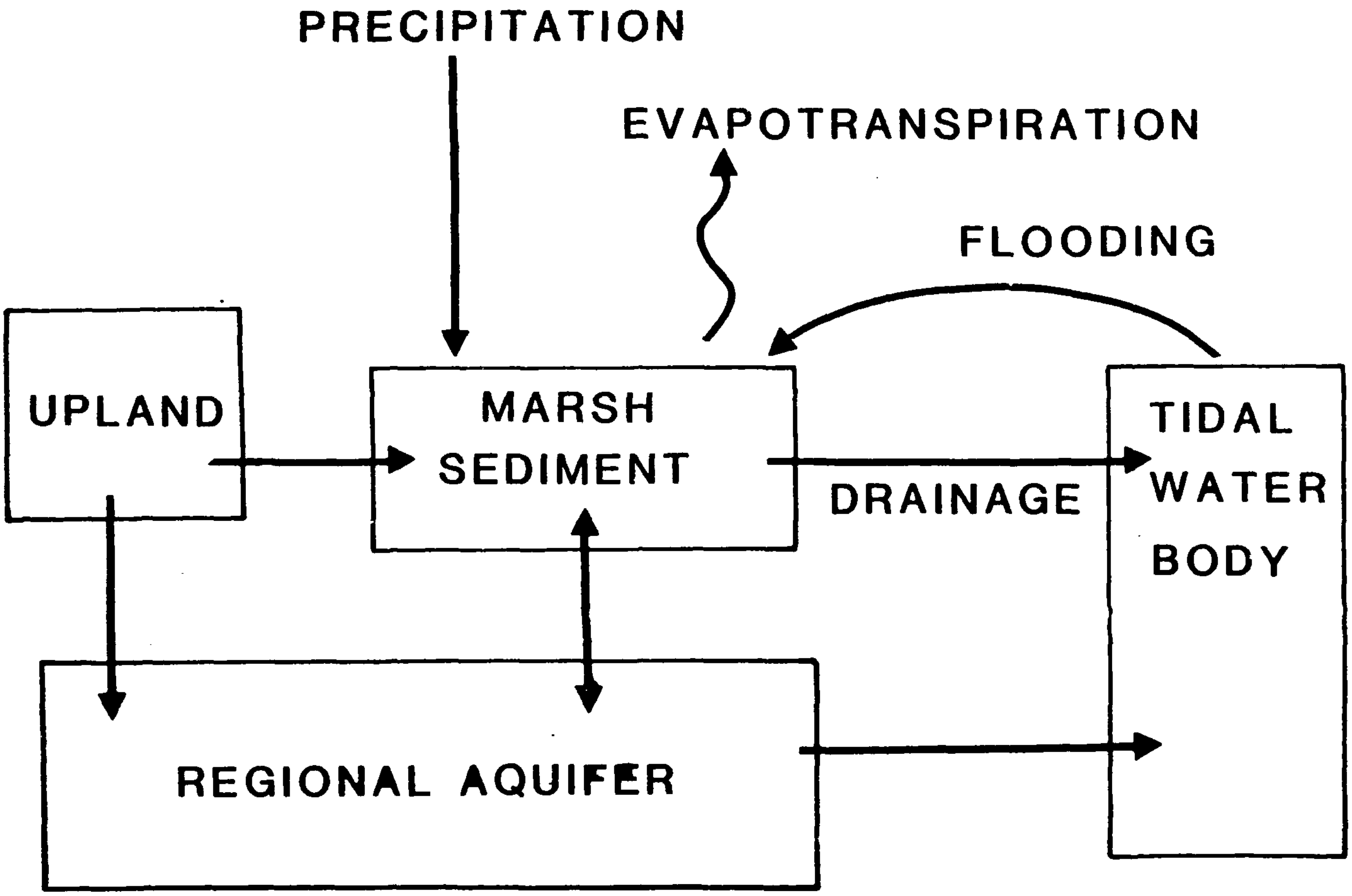

Figure 1. Elements of the water budget in a salt marsh

Thirteen piezometers were installed to a depth of 50 cm in an array perpendicular to a creekbank. The piezometers were constructed of 1/2" (nominal) schedule 40 PVC pipe, slotted over 15 cm at the lower end. The time constant for recovery of the water levels during a slug test was on the order of two hours, a small duration with respect to day-to-day variations in the head of interest here. Observations of hydraulic head were made approximately every other day during the two month period August-September 1984.

Evaporation and precipitation were estimated from meteorological data recorded by the National Weather Service observatory at Logan International Airport, 3 km southwest of the study site. Evapotranspiration was estimated from the net heat flux to the marsh surface by the method of Priestly and Taylor (1972), which applies to extensive areas of well-watered grassed surface (Brutsaert, 1982). A full description of the methods used to estimate the net heat flux to the marsh canopy from the meteorological record is given by Nuttle and Hemond (1988).

Carter Creek Marsh

The Carter Creek marsh (37°20'N 76°35'W) is located within the lower York River, Virginia, a sub-estuary of the Chesapeake Bay. The sediment is less than 1 m thick and consists of a mixture of silt, clay and organic matter (16% ash-free dry weight, Harvey et al, 1987). Beneath this is a relatively impermeable layer of sand intermixed with fine material of mineral and organic origin. The majority of the marsh is low marsh, vegetated by Spartina alterniflora of medium height (1 m). A narrow high marsh fringe occurs at the base of an upland hillslope and is dominated by Distichlis spicata and Scirpus robustus. Typha angustifolia, Rumex verticillatus, and Polygonum sp. are present at the marsh-upland boundary. The mean and spring ranges of the astronomical tide are 80 and 100 cm respectively, and the low marsh is inundated approximately 340 days per year.

Piezometers at Carter Creek were constructed from 1 cm I.D. acrylic tube screened over their lower 10 cm and capped with conical acrylic tips.

Wells were similarly constructed, but screened over their entire length. Wells and piezometers were installed from an elevated catwalk in nested arrays at five locations across the marsh surface. At each location, piezometers were installed to three depths; 25 cm, 45 cm and 75 cm. Three additional wells were installed along the transect between stations 3 and 4. The average response time for the piezometers was 8.5 min, which indicated that instrument response to changing head conditions in the marsh was rapid enough to follow changes induced by semi-diurnal flooding of the marsh. Piezometers with response times greater than 20 min were not used. Hydraulic heads were measured every 15 min over complete tidal cycles on July 11, July 26, August 11, and September 9. 1985. Evaporation was measured using a Class A pan located in the marsh.

MODEL FORMULATION

Governing Equation for Head

Consider a prismatic volume taken from the layer of marsh sediment and in which a piezometer is used to measure the locally-averaged hydraulic head, h.

$$h = \frac{P}{\gamma_w} + z \qquad (1)$$

in which P is the average pore pressure, γ_w is the unit weight of water and z is the elevation of the piezometer screen. Changes in hydraulic head over time are related to changes in the water content of the sediment due to evaporation, q_E, infiltration, q_I and lateral water fluxes within the sediment, q_H. Lateral fluxes are related to the spatial distribution of head by Darcy's law;

$$\overline{q_H} = -K \frac{d\overline{h}}{dx} \qquad (2)$$

If the depth and the hydraulic properties of the sediment are assumed to be constant and uniform throughout the marsh and if water movement in the sediment occurs predominantly perpendicular to the creekbank, then the distribution of hydraulic head is controlled by the following differential equation:

$$\frac{\delta h}{\delta t} = \frac{K}{S_S} \frac{\delta^2 h}{\delta x^2} + \frac{1}{DS_S}(q_I - q_E) \qquad (3)$$

in which x is distance from the creek, K is the hydraulic conductivity and S_S is the specific storage of the sediment.

Solution for the Distribution of Hydraulic Head

Solutions of Equation 3 for the distribution of hydraulic head, subject to the boundary conditions and sediment properties of each marsh, constitute models of the subsurface hydrology. In Belle Isle marsh, the marsh surface is infrequently flooded by the tides, and a closed form solution of Equation 3 could be found for the non-flooded periods, zero infiltration. The development of this solution is given in detail by Nuttle (1988). Daily flooding of the marsh surface in the Carter Creek marsh necessitated the use of a numerical solution. An existing finite difference code (Wang and Anderson, 1982) was modified to account for the effects of tidal flooding on hydraulic head in the sediment. The details of this model are described in Harvey et al (1987). Evapotranspiration was excluded from the model developed for Carter Creek because it plays a relatively small role in the response of hydraulic head during the short periods of time in which the sediment surface is exposed to air.

MODEL CALIBRATION

The water conductance and storage properties of the sediments of the two marshes, required by the models, were determined both by in situ and by laboratory techniques (Nuttle, 1988; Harvey et al, 1987). The values of the sediment parameters were not adjusted to improve the fit between the modeled and the observed head distributions. The sediment in Belle Isle marsh has a hydraulic conductivity of 1.7×10^{-4} cm/s and a specific storativity of 9.4×10^{-4} 1/cm. The sediment in Carter Creek marsh has a hydraulic conductivity of 7.4×10^{-4} cm/s. A depth integral of the specific storage, the specific yield, was used as the storage parameter in the Carter Creek model, and its value was found to be 0.032.

OBSERVATIONS AND COMPARISON WITH MODEL RESULTS

Belle Isle Marsh

The water content and hydraulic head in the sediment of Belle Isle marsh follows a cycle driven by the spring/neap tide cycle, Figure 2. When high tides exceed a threshold (10 ft relative

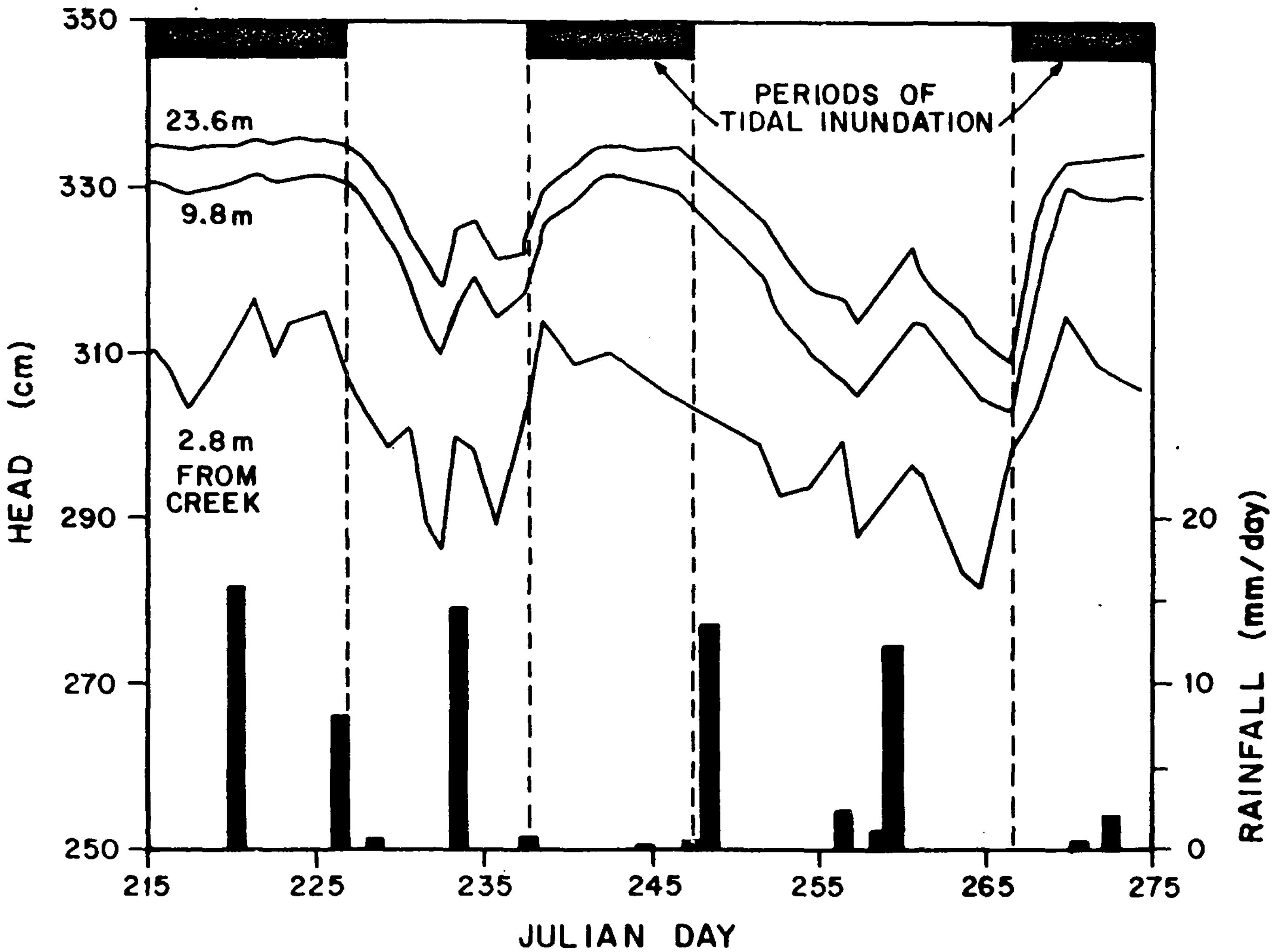

Figure 2. Daily flooding of the marsh by the highest spring tides maintains the hydraulic head at levels near the marsh surface. Hydraulic head declines as a result of a net loss of water from the sediment by evapotranspiration during the intervening dry periods. Periods of rainfall cause a temporary recovery in head. Surface elevation and proximity to a tidal creek are factors in the head differences between piezometers.

to low water in Boston Harbor) the entire surface of the marsh floods and water infiltrates into the sediment. A period of daily flooding drives the water table to an equilibrium position near the marsh surface. As long as daily tidal flooding continues, the water table is held at this position and the water content of the sediment remains constant. During periods of neap tides, in which the marsh surface is not flooded, there is a net loss of water from the sediment by evapotranspiration and drainage near a creekbank, and hydraulic head decreases. On the return of the spring tides and daily flooding of the marsh surface, infiltration replenishes the water content of the sediment, and the cycle begins again. The amount of infiltration that can occur during a period of daily flooding is determined by the amount of water lost from the sediment in the preceding non-flooding period. Precipitation occurs at random with respect to this cycle and affects the net water loss from the sediment during the non-flooding periods. However, the largest fluctuations in water content are driven by the cycle of flooding and non-flooding periods driven by the spring tides.

The model for the head response during non-flooding periods reproduces the general distribution of head throughout a period of net water loss, Figure 3. The uppermost curve in Figure 3 represents the fitted initial head distribution, which is controlled by the morphology of the marsh surface. The other curves are model predictions. In the interior of the marsh, greater than 15 m from the creekbank, there is virtually no lateral water movement

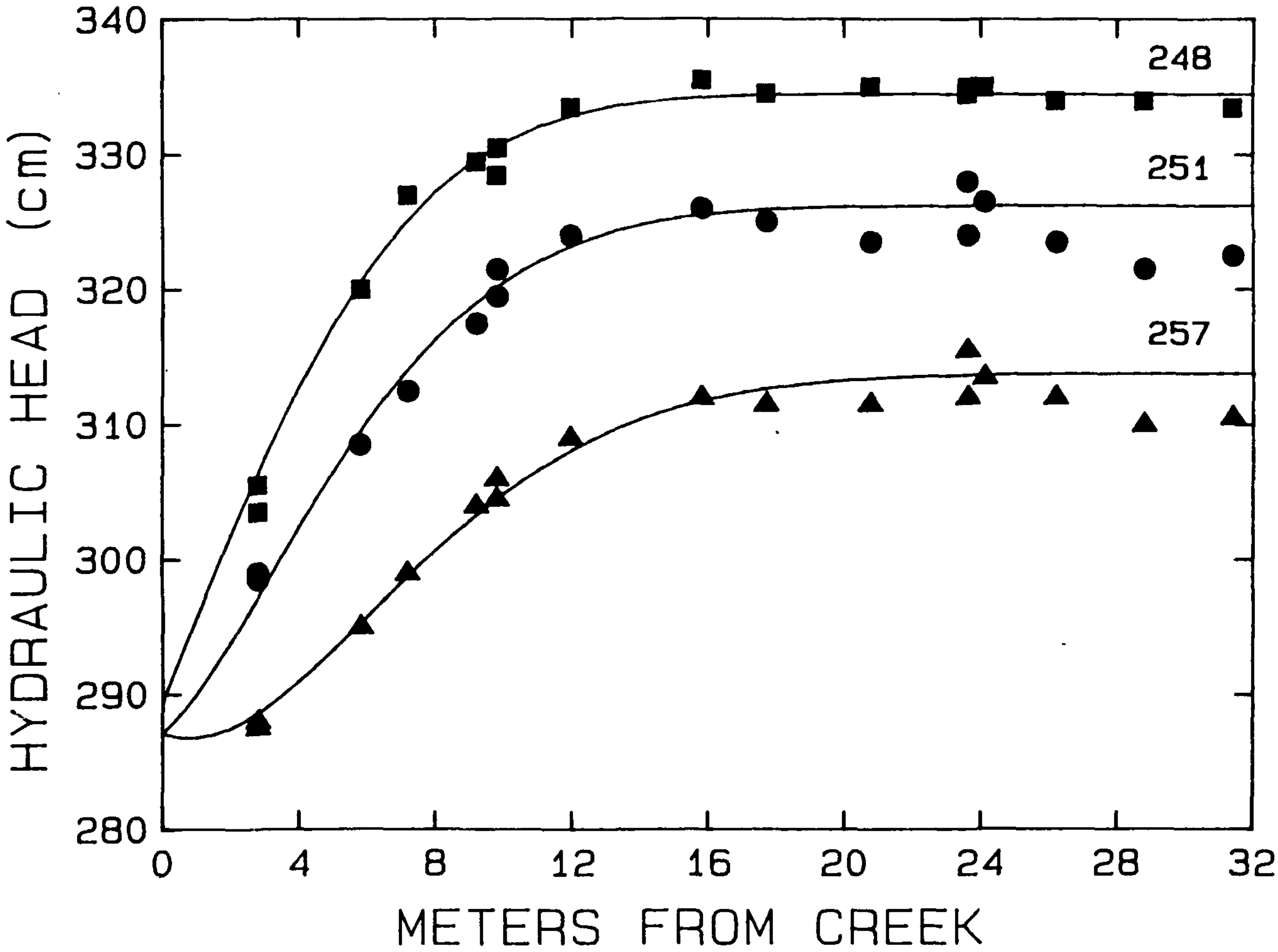

Figure 3. The model of head response is fitted to piezometer observations during a period of net water loss from the sediment due to evapotranspiration and drainage. The uppermost curve is the fitted initial head distribution. The other curves are model results for the Julian dates shown.

towards the creek. Here, in the absence of any significant exchange with an underlying aquifer, all water that enters the sediment is eventually removed by evapotranspiration. A transition zone lies between 2.5 m and 15 m from the creek in which lateral water movement is driven by alternating periods of daily surface flooding and periods of no flooding, associated with the spring-neap tide cycle. Within 2.5 m of the creek bank is a region in which semi-diurnal fluctuations in the water level in the creek drive an oscillatory lateral water flux with an amplitude of 2×10^{-4} cm/s at the creekbank (Nuttle, 1988).

The one-dimensional model of head response to water loss by evapotranspiration and drainage can be used to estimate the magnitude of the lateral flux drainage towards the creek (Nuttle, 1988). Mean horizontal fluxes in the region of 0-8 m are on the order of 6×10^{-6} cm/s for the long 10-15 day non-flooding periods. Higher fluxes, 5×10^{-5} cm/s, occur at the creekbank in the first day of drainage. Generally, drainage fluxes are highest for short periods of drainage and frequent surface flooding, but the high lateral fluxes occur only in regions close to the creek bank. Long periods of drainage result in smaller mean fluxes distributed over larger areas of the marsh.

Carter Creek Marsh

Changes in hydraulic head and tide stage over a complete tidal cycle in Carter Creek marsh are shown in Figure 4. Hydraulic head, water table height, and elevation of the marsh surface are referenced to a common datum in this figure and can be compared directly. There are three stages in the response of head at any point. Stage 1: on the receding tide, hydraulic head at a point in the sediment falls with the tide until the surface above the point is exposed. Stage 2: Heads continue to decline, but at a lower rate than the rate of change of water levels in the creek. Hydraulic heads in the sediment are higher than in the creek, so water drains from the sediment into the creek, resulting in the decline of head in the sediment. Stage 3: As the sediment surface re-floods, the head in the sediment under the flooded portions of the sediment is equal to the level of the overlying water, after which heads vary with tide level. Note that when the marsh is

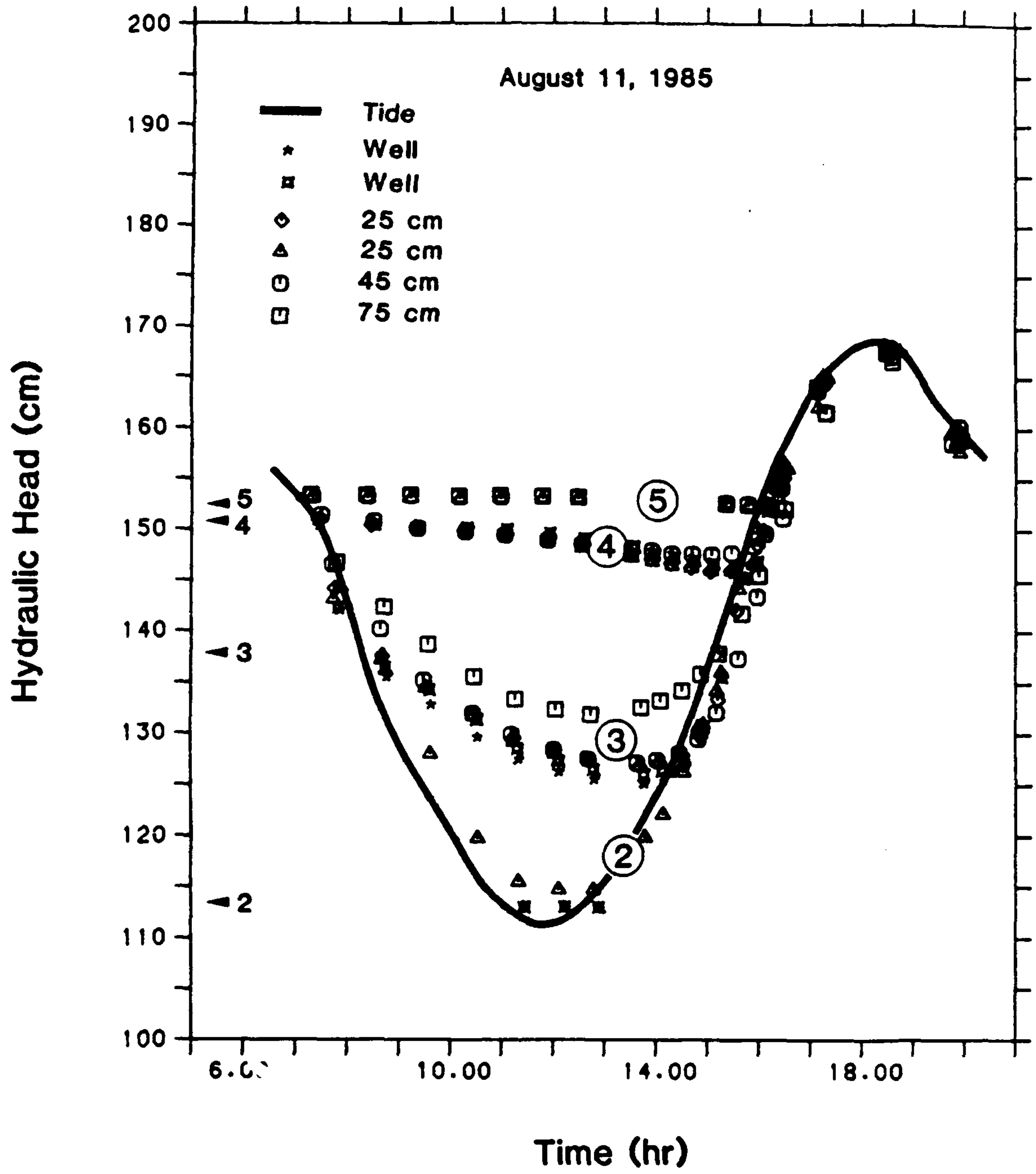

Figure 4. Changes in tidal stage and hydraulic head over a complete tide cycle in Carter Creek marsh, August 11, 1985. Station data are numbered and circled. Station 2 is located at the base of the creekbank and stations 3, 4,5 are 1, 4.5, and 8 m from the creek bank. Numbered arrows on the ordinate point to the marsh surface elevations for each station.

completely flooded, lateral gradients in head are essentially zero and no subsurface flow occurs. When the marsh surface is exposed, gradient in hydraulic head is controlled by the gradient in the elevation of the marsh surface.

Water remained ponded behind the levee on the marsh flat throughout the falling tide. The elevation of the ponded surface dropped an average of 6.8 mm during the period of tidal exposure over the four monitored tidal cycles. Comparison with pan evaporation rates indicates that most of the water loss occurs as infiltration

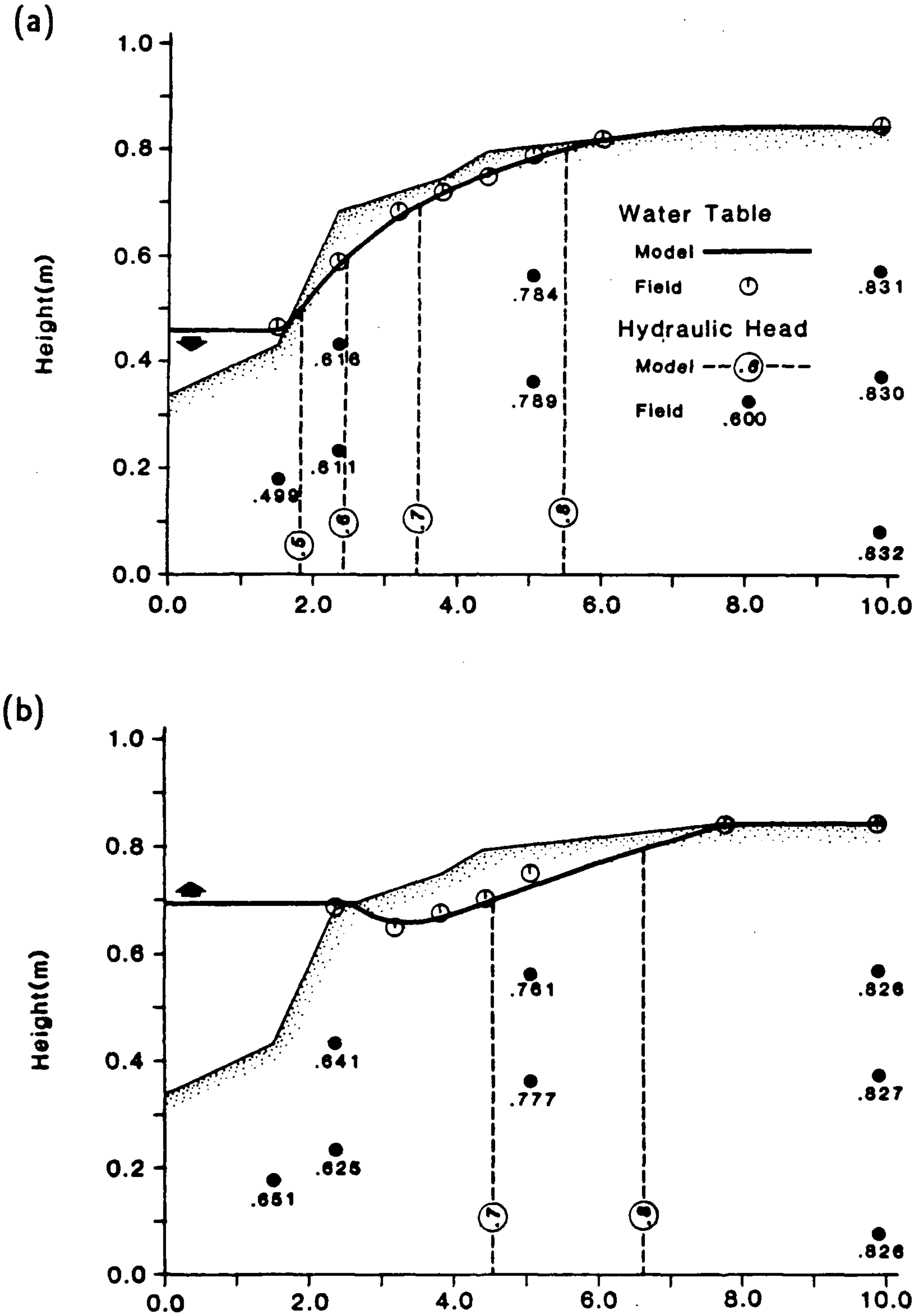

Figure 5. Comparison of the measured and modeled hydraulic head distribution in Carter Creek on 8/11/85. (a) Falling tide; (b) Rising tide. The solid line beneath the marsh surface represents the predicted height of the water table. Dashed lines are predicted equipotentials. Symbols represent measured heights of the water table and hydraulic heads at depth (Key – *upper right*).

and drainage through the sediment.

Agreement between model and field data is excellent early in the tidal cycle but deviations increased over time (Figure 5). Nevertheless, differences between observed and predicted head values were small (< 3 cm) in a flow system that was 80 cm deep. The overall qualitative behavior of the flow system was well represented by the model. For example, bi-directional flow during the rising tide (i.e. recharge to the sediment at the creek bank, continuing discharge at interior regions) was correctly predicted (Figure 5).

Modeled pore-water discharges were calculated at the creek bank over four complete tidal cycles. Discharge velocities were 3 X 10^{-5} cm/s, averaged over a complete tidal cycle. Model simulations indicated that horizontal replacement of pore water by inflow at the creek bank was negligible. Vertical infiltration and lateral discharge from the interior marsh replaced approximately 70% and 30% of the volume of water drained at the creekbank, respectively.

CONCLUSIONS

Based on our model results, we conclude that the magnitude and extent of lateral drainage in salt marsh sediments is controlled by four factors; 1) the morphology of the marsh surface, 2) the time between successive tidal flooding events, 3) the hydraulic properties of the sediment, and 4) the rate of evapotranspiration. The patterns of lateral water movement associated with two fundamental cycles in the hydrologic regime of Belle Isle marsh have been examined: semi-diurnal fluctuations in creek level and the spring-neap cycle of flooding and non-flooding periods. In each case, higher rates of infiltration and lateral water fluxes are found close to the creek bank. Geomorphological variables exert the most control over the volume of pore water exported at the creekbank, and thus on the turnover rate of pore water near the creekbank. The geomorphology of the marsh surface has two effects; 1) the topography of the surface controls the initial distribution of hydraulic head following the flooding and complete saturation of the sediment by tides, and 2) the overall elevation of the sediment surface relative to the range of tides determines the frequency of complete tidal flooding of the marsh. Hydraulic properties of the sediment are next in importance in determining turn-over rates in the sediment. The overall pattern of high turn-over rates near the creekbanks is determined by the combination of all factors, so that the drainage velocities in Belle Isle marsh and Carter Creek marsh are very similar even though the geomorphology and hydraulic characteristics of the sediments differed.

Our results highlight the importance of the geomorphology of marsh sediments in maintaining the ecological functions of coastal marshes. Impacts can result from direct changes in the morphology of the marsh by filling or dredging and indirectly from changes that may affect the characteristics of the tides. We have also identified the creekbank region in coastal marshes as an area of dynamic subsurface hydrology. Higher rates of infiltration and drainage contribute to higher rates of pore water turn-over and air-entry into the sediment that affect microbial processes controlling water quality in the marshes and estuaries.

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Hydrology of Arctic Wetlands

Scott B. Robertson
ARCO Alaska, Inc.

INTRODUCTION

The image that comes to mind when most people think of wetlands is usually one of cypress filled swamps or coastal salt marshes and inland freshwater marshes and prairie potholes dominated by herbaceous vegetation. A great deal has been written describing the structure and ecology of these systems (e.g., Good, Whigham, and Simpson, 1978; Weller, 1981; OTA, 1984; Mitsch and Gosselink, 1986). Most concern and interest has been focused on these types of wetlands because they are the ones most frequently encountered by people and have been the most severely impacted by man. It has been estimated that up to 50% of the original wetlands in the conterminous United States have been lost or altered (OTA, 1984). However, huge areas of relatively untouched wetlands can still be found in Alaska. Perhaps more than 75% of all remaining United States' wetlands exist in Alaska (data from ASIWPCA, 1984, cited in Mitsch and Gosselink, 1986). Arctic wetlands are similar in appearance to many in the lower-48 states but on closer examination are found to be quite different in formation and function from more temperate wetlands. One aspect of the uniqueness of arctic wetlands is their hydrology. The focus of this paper is the description of how hydrology relates to the other unique properties of wetlands on the North Slope of Alaska.

TEMPERATURE

As should be expected, freezing temperatures exist for most of the year (Fig.1). Average temperatures above freezing occur in only three or four months. During the summer, monthly means rarely exceed 10°C but daily highs have exceeded 27°C. Mid-winter monthly means are frequently lower than -30°C and daily lows have exceeded -50°C. Wind chill can depress the apparent temperature by an additional 30°C or more (ARCO records from Prudhoe Bay).

Thousands of years of these temperatures has resulted in permafrost that reaches as deep as 660 m below the surface (Gold and Lachenbruch, 1973). During the summer the surface of the ground thaws, typically to a depth of 0.5 m by the end of the summer. This alternately thawing and freezing ground is called the *active layer* and is the zone in which (during the summer) liquid water can exist. Thus, permafrost affects hydrology by limiting the downward percolation of water.

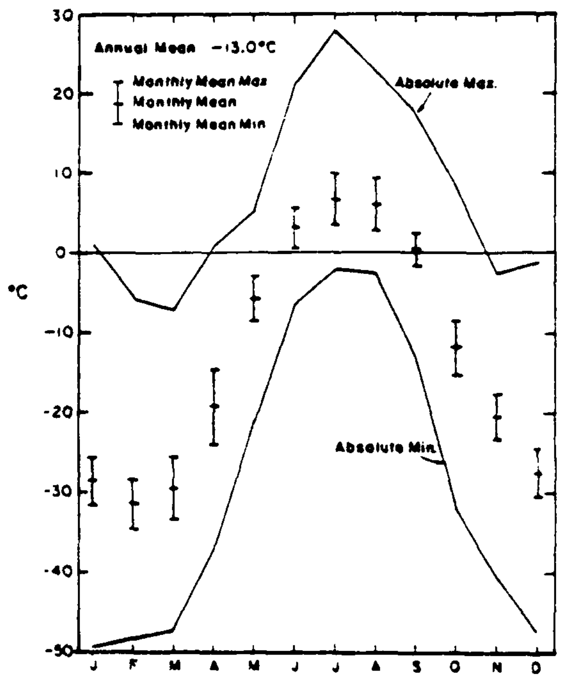

Figure 1. Annual distribution of monthly ranges of air temperature for the Prudhoe Bay region based on an eight year record, 1970-77 (Walker et al, 1980).

PRECIPITATION

Excessive amounts of precipitation are not what have led to the formation of wetlands in the Arctic. On the contrary, comparison with precipitation rates in the vast arid areas of the western contiguous United States (<10 in/yr, Fig.2) demonstrates that the North Slope of Alaska (Fig. 3) could qualify for classification as a desert. Precipitation along the Beaufort Sea coast (e.g., Barter Island, Prudhoe Bay, Barrow) averages approximately 180 mm (7 in.) (Dingman et al, 1980; Walker et al, 1980).

An additional aspect of the problems of providing water to North Slope wetlands is that 65% of the precipitation is in the form of snow

(Walker et al, 1980). When it melts in late spring ("breakup") the ground is still frozen so most of the water drains off the flat terrain as sheet-flow to the streams and rivers and is unavailable to the wetlands.

Because the terrain is quite flat (most of the landscape is classified as either flat or gently rolling plain) over much of the North Slope, minor depressions are able to hold the remnants of the breakup flows. Additional precipitation in the form of light rain and drizzle occurs in the summer but water loss by evapotranspiration during this period is of a similar magnitude (Bunnell et al, 1975). Thus, although precipitation levels are low and water losses from breakup flows over frozen ground and from evapotranspiration are relatively high, wetlands can exist in the Arctic because the absence of much vertical relief limits the horizontal movement of water and the presence of permafrost limits the downward movement of water. The result is a complex of seasonal

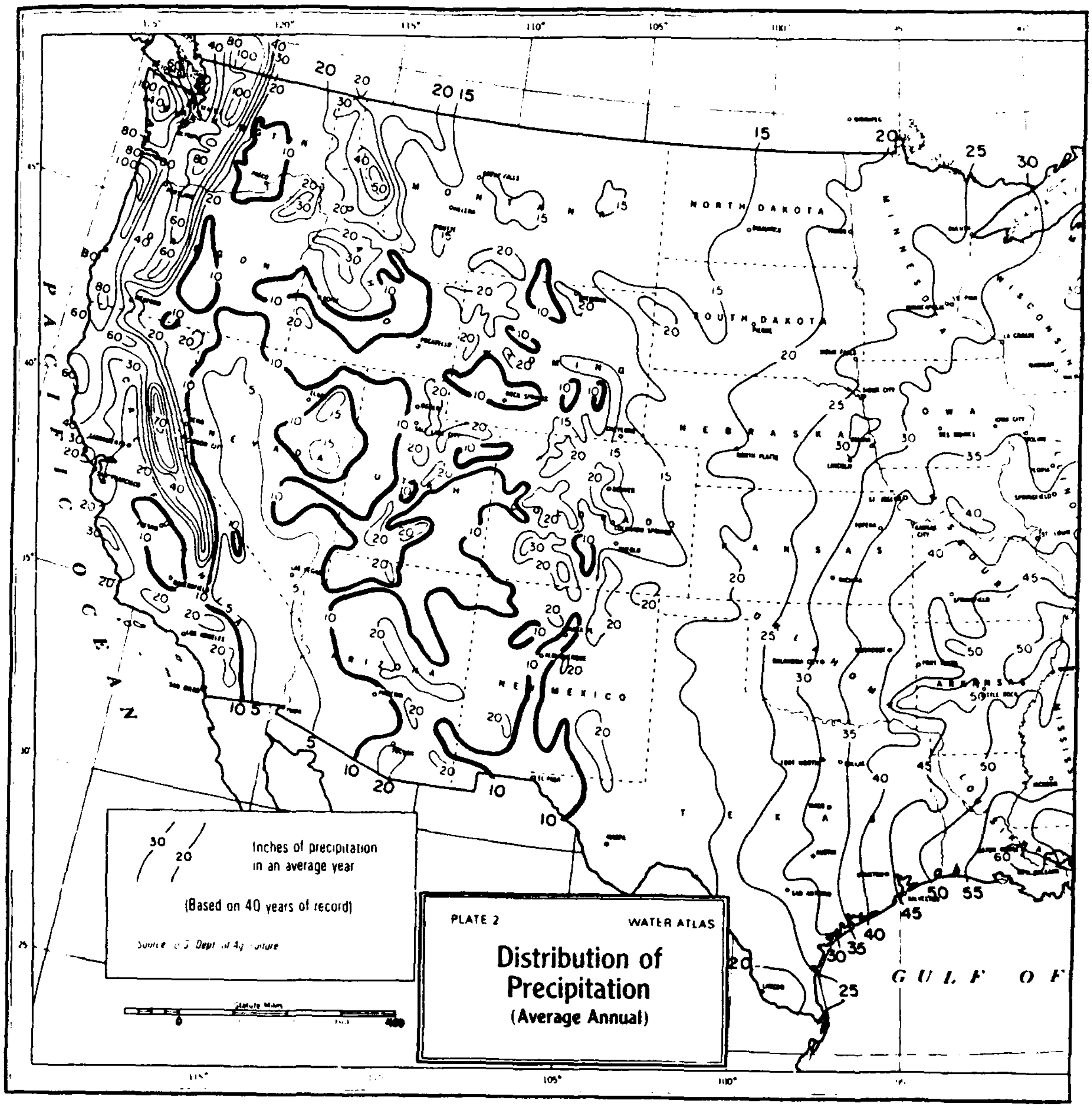

Figure 2. Distribution of precipitation in the contiguous United States (Geraghty et al, 1973, used by permission).

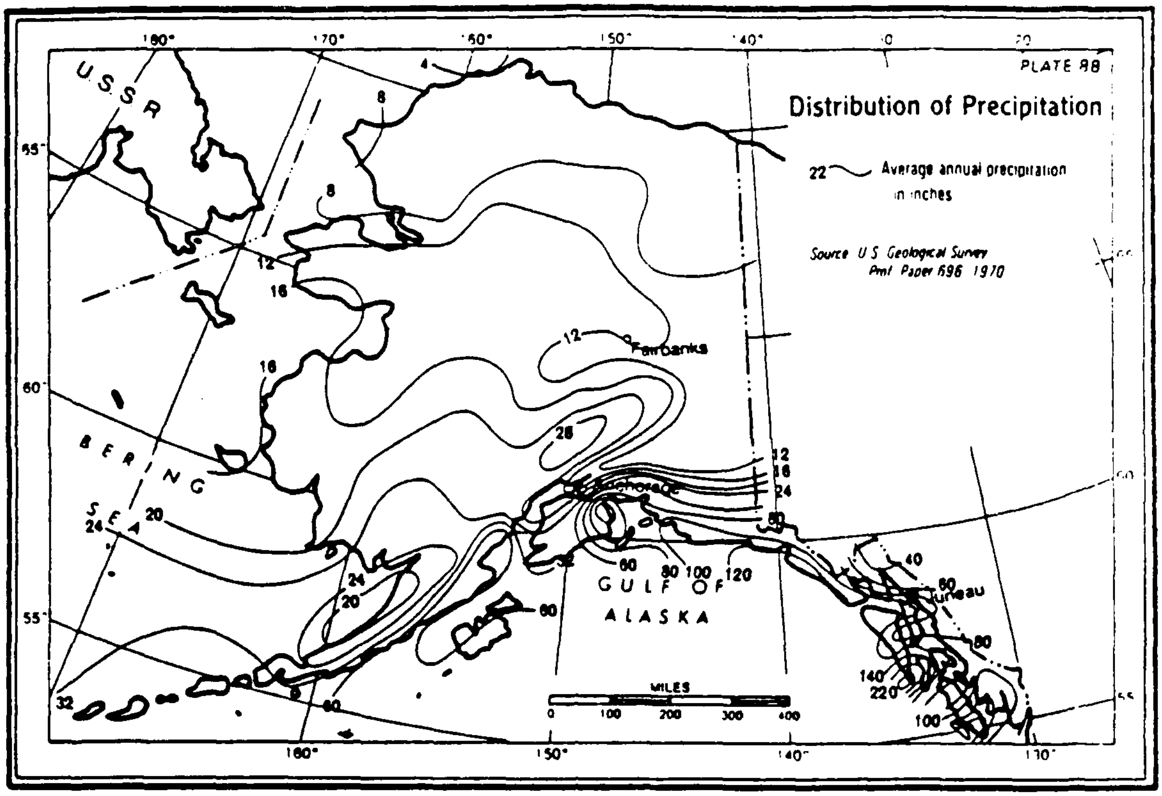

Figure 3. Distribution of precipitation in Alaska (Geraghty et al, 1973, used by permission).

thaw-ponds, shallow emergent wetlands, deep open lakes, and partially flooded drained lake basins; all interspersed among dry and moist tundra habitats.

RIVER HYDROLOGY

The flow pattern of North Slope rivers are primarily a function of breakup and summer rain. Flow during the winter is virtually zero. At breakup the sudden input of meltwater brings river flow to a peak. Duration of this peak and subsequent flow patterns during the remainder of the summer are functions of the nature of the river.

The Kuparuk River is primarily a tundra river. Almost all of its drainage area is on the North Slope and is 9477 km^2 (3659 mi^2) in size (Selkregg, 1975). Less than 10% of its drainage lies above 600 m elevation (Drage et al, 1983). Break-up flow of the Kuparuk River is quite high but diminishes rapidly after a couple of weeks (Fig.4). Flow rates are relatively low throughout the remainder of the summer with periodic peaks in response to individual rain events. Increased rainfall toward the end of summer increases the average flow. With the onset of freezing temperatures in September, flow begins to steadily decrease. Under-ice flow is present through November. River ice ultimately reaches a thickness of approximately 2 m (6 ft) where water depth permits.

The Sagavanirktok River has its origins in the Brooks Range with approximately 50% of its drainage above 600 m elevation (Drage et al, 1983). As a mountain based river it has a flow pattern quite different from that of the Kuparuk River (Fig.4). Although its drainage area is larger, 14,364 km^2 (5,546 mi^2) (Selkregg, 1975), its breakup flows are lower. However, as a result of more frequent rain in the mountains, flow during the summer is maintained at a higher level. The flow pattern in the fall is similar to the description given for the Kuparuk River.

STREAM HYDROLOGY

Flow in tundra streams is somewhat similar to the pattern seen in the rivers, although the flow volumes are substantially lower (reflecting the smaller drainages: 72 to 148 km^2) and the temporal distribution is dramatically more extreme. Typical breakup events of coastal plain streams last 10 to 12 days (Fig.5) (Drage et al, 1983). Examination of four streams west of the Kuparuk River show that post breakup discharge rates are very low and, in lower than average rainfall years, may reach zero (Fig.6) (PN&D, 1985).

After breakup most tundra streams become

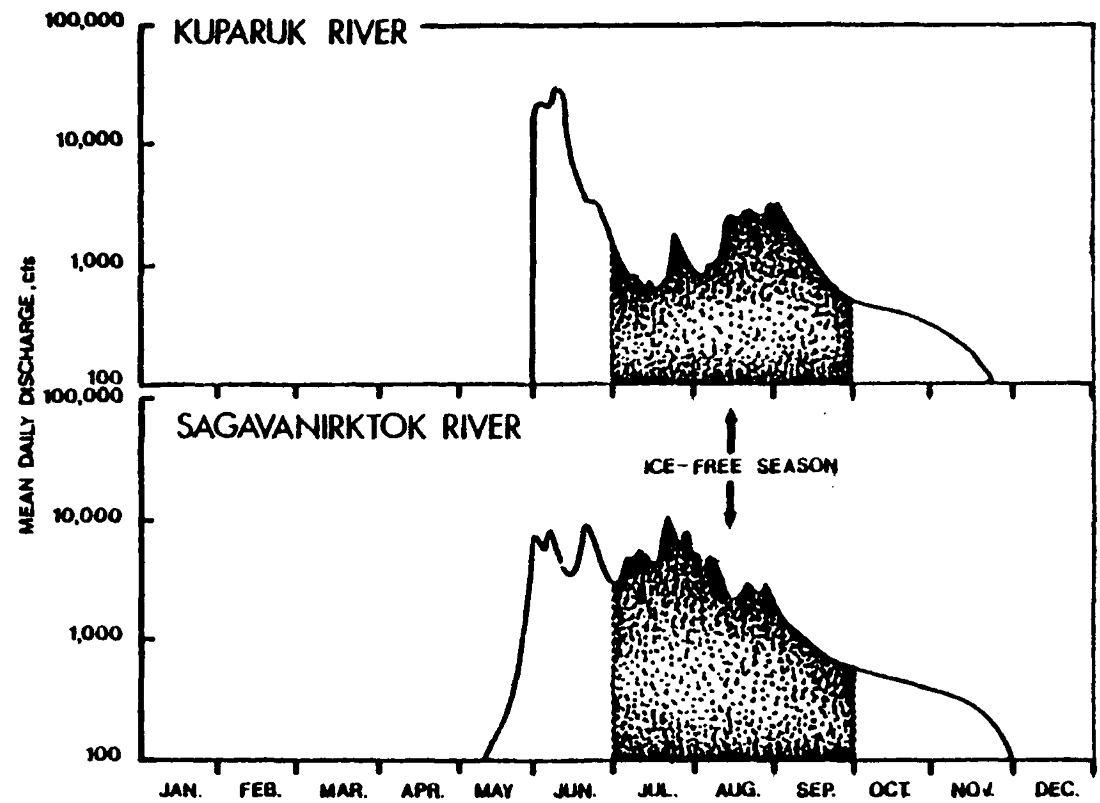

Figure 4. Annual discharge rate distribution of the Kuparuk and Sagavanirktok Rivers. (Adapted from Carlson et al, 1977 by Griffiths and Gallaway).

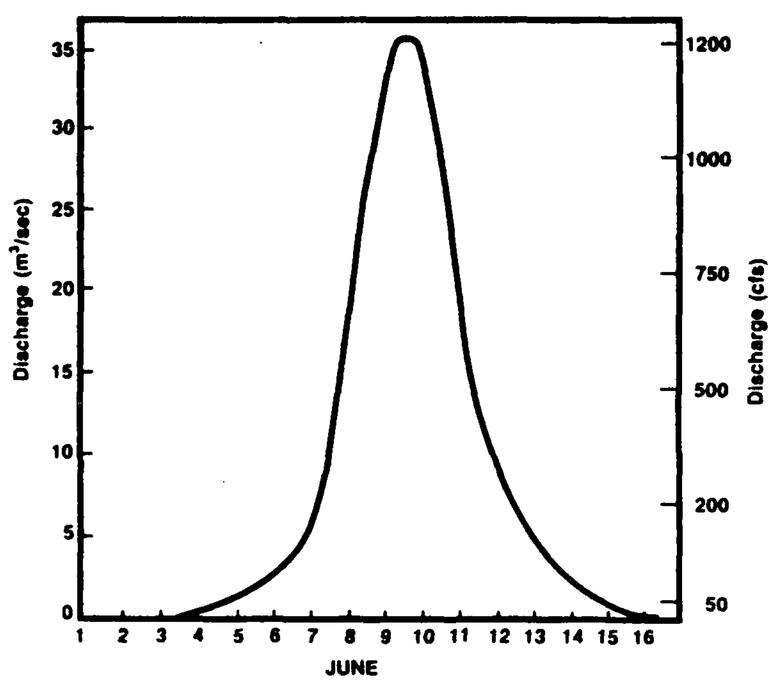

Figure 5. Discharge rate of the Sakonowyak River during 1981 breakup (Drage et al, 1983).

merely a series of ponds with little or no flow through the narrow stream beds that connect them. Late summer rainfall can increase flow rates slightly. The fact that these streams can have no flowing water despite the fact that they are surrounded by wetlands is a function of the hydrology of the soil. In other sites on the coastal plain, runoff equivalent of 5% of summer rainfall has been measured (Brown et al, 1968; Lewellen, 1972, cited in Miller et al, 1980). For runoff to occur the ground must first become saturated. Overflow and runoff from tundra ponds may not generally occur unless the summer's rainfall is well above average (Miller et al, 1980).

SOIL HYDROLOGY

"Groundwater" is present only in the thawed active layer. This soil is generally rich in organic material and is therefore very porous and permeable. Movement of this water is limited because the topographic gradients are so slight. Flow is typically downward through the active layer in elevated (dry) sites, horizontal in transition areas between dry and wet sites, and practically stagnant in depressed (wet) sites (Fig.7) (Ryden, 1981). The polygonal nature of much of the landscape also lends itself to limiting horizontal flow by the presence of the many small catchment basins (Miller et al, 1980). The

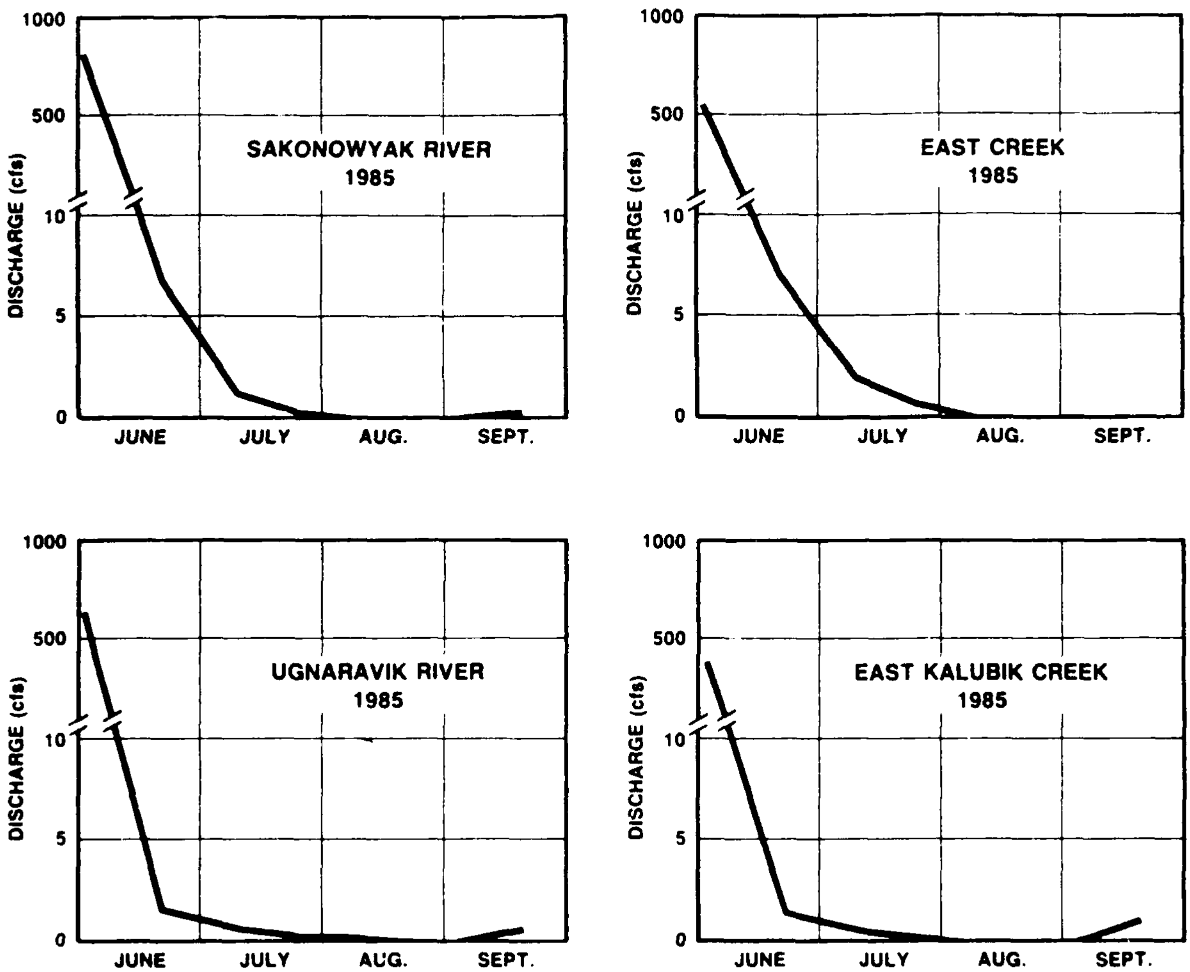

Figure 6. Discharge rates of four drainages during 1985. The drainage areas upstream of the summer measurement stations were Sakonowyak River: 148 km^2 (57 mi^2), East Creek: 132 km^2 (51 mi^2), Ugnaravik River: 85 km^2 (33 mi^2), and East Kalubik Creek: 72 km^2 (28 mi^2). Note the two-orders-of-magnitude change in scale on the discharge axis. Precipitation during most of this summer was below average (data from Pn&D, 1985).

subsurface depth of these basins is more pronounced than the surface appearance would indicate (Fig.7). The thermal conductivity of saturated peat can be two to three times greater than that of well drained peat (Ryden and Kostov, 1980). This will result in the active layer being thicker under wet depressions than under dry, elevated sites. Thus, the catchment basins formed by permafrost will have more relief than seen at the surface. The wetlands and isolated ponds and lakes therefore contribute little to the mid-summer water budget of tundra streams. Most of the "groundwater" contribution to the tundra streams is from the well drained slopes alongside the stream.

soil which can occur as horizontal lenses and vertical wedges. Wedges are formed by the freezing of water that fills cracks in the ground that result from thermal expansion and contraction during the seasonal freezing and thawing of the ground. Summer thawing allows additional water into the thermal shrinkage cracks, which subsequently freezes and enlarges the wedge. The heaving of the ground over the edges of the wedge creates narrow raised ridges that are drier than the surrounding ground. These are frequently organized into a polygonal network. Low-centered polygons may have wet adapted vegetation internally and dry adapted vegetation on their edges even though the difference in elevation may be less than 0.5m.

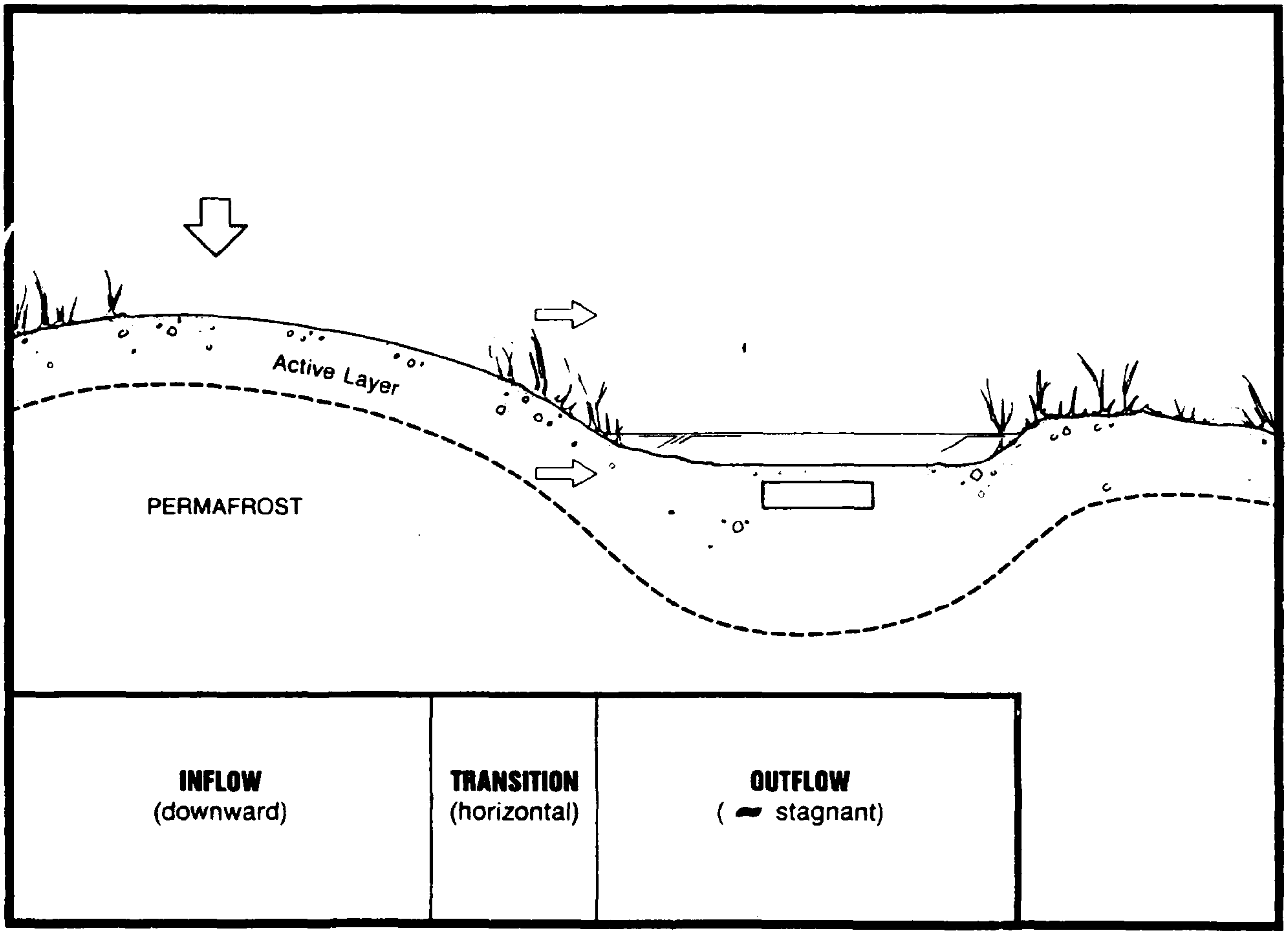

Figure 7. Schematic of water flow in permafrost-based wetland soils. Vertical elevation differences are approximately to scale, the difference between dry, elevated areas and pond levels may be less than 0.5 m. Horizontal differences are not to scale and may vary widely.

GEOMORPHOLOGY

The landscape of the Arctic is a mosaic of a variety of landforms, moisture regimes, and vegetation. Much of this is a function of ice in the

Troughs within the edges of low-centered polygons may remain wet enough that subsidence from thermal degradation of the

underlying permafrost occurs (thermokarsting). This can result in the transformation to high-centered polygons. Because edges of high-centered polygons are low and are interconnected, these areas are well drained and, except for edges, support dry adapted vegetation.

Other landforms in arctic tundra include stangmoor, which is found in very wet areas and consists of discontinuous ridges similar to the edges of low-centered polygons. Hummocky terrain (up to 0.5 m diameters) is common on well drained slopes. Where there is little ice in the soil as in some well drained areas or where there has been relatively little exposure to repeated frost cycles, such as in recently drained lake basins, non-patterned ground may be present.

Drained-lake basins are one part of a thaw-lake cycle which is unique to permafrost areas (Britton, 1967). An area of low-centered polygons can become wetter as the collection of water in the centers results in thermokarsting. The more water present results in more subsidence which results in collection of more water in a positive feedback. The melting of ice wedges will result in the polygons coalescing and ponds becoming larger. Continued thermal erosion at the edges, subsidence, and capture of adjacent polygons ultimately results in the formation of a large, shallow lake. Eventually a low divide may be breached or a stream captured and the lake will drain. If the lake drains only partially, exposed land areas will refreeze and heave upward, creating a complex of islands. A completely drained lake may have a relatively uniform appearance at first, but eventually, as the ground freezes and thaws, reformation of low-centered polygons occurs, completing the cycle. The process may take several hundred to a few thousand years to complete the cycle.

The permafrost-based nature of Arctic wetlands results in the absence or extreme limitation of most of the valuable functions expected of wetlands (Robertson, in press). Habitat for wildlife is a notable exception to this generality. Morphologically diverse drained-lake basins, containing lakes with complex shorelines and islands are frequently excellent habitat for waterfowl, especially for nesting geese such as black brant. The many kinds of tundra habitat on the North Slope support large numbers of migratory water birds during the summer, but densities are relatively low. Other Alaska wetland areas, such as the Yukon-Kuskokwim Delta and the Yukon Flats support densities and numbers that are two orders of magnitude greater than the North Slope (Derksen et al, 1981).

CONCLUSION

That Arctic wetlands are in a continual state of evolution is not particularly unique; that ice in the soil is an important factor in that evolution is unique. This has ramifications in terms of mitigation. The petroleum industry on the North Slope of Alaska has mitigated the effects of their development primarily by avoidance and minimization (Smith and Robertson, 1988). As a consequence of this, drill pads are seldom placed in their ideal locations (from a reservoir development standpoint) and gravel roads rarely go in straight lines. Rather, they snake around to avoid valuable wetlands in the mosaic of wetlands that exists.

Other forms of mitigation, such as the enhancement or creation of wetlands, generally involve changes to the hydrology of an area. For this reason anything one does to enhance or create wetlands in the Arctic will have thermal consequences. Thus, because of effects on ice and permafrost, whatever manipulations are employed are likely to have ultimate consequences very different from what was originally planned. It is probable that techniques developed in the lower 48 will be quite inappropriate in the Arctic. Mitigation in the Arctic must rely on minimization and avoidance.

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Groundwater Transport between Hillslopes and Tidal Marshes

Judson W. Harvey, Randolph M. Chambers, William E. Odum
Department of Environmental Sciences
University of Virginia

INTRODUCTION

Most studies of groundwater discharge to surface waters in the United States have been conducted in lakebeds (Winter, 1983; Munter and Anderson, 1981; Lee et al., 1980; McBride and Pfannkuch, 1975) or in small headwater streams in steep catchments (Stephenson and Freeze, 1974; Dunne and Black, 1970). In wetlands, studies of groundwater discharge have been concentrated in the peatlands and prairie potholes of the northern Midwest (see reviews in Heinselman, 1970; Hubbard, 1981 and Mitsch and Gosselink, 1986). Groundwater research in wetlands situated along coastal rivers or estuaries is vastly under-represented in the hydrological literature. Perhaps this is due to a perception that subsurface flow should account for a negligible percentage of the total hydrological transport in those systems. Yet the documented decline in water quality in highly productive and commercially important estuaries such as the Chesapeake Bay has been linked to non-point discharges of pollutants which include groundwater discharge (Kuo and Younos, 1986). Several investigators have considered the importance of groundwater discharge to estuaries as a mechanism for the transport of nutrients and pollutants from upland sources to already stressed and eutrophic surface waters (Gilliam and Skaggs, 1986; Peterjohn and Correll, 1984; Valiela et al., 1980; Lee, 1980; Valiela et al., 1978). Of these, only Valiela and his colleagues attempted to characterize the interaction between upland groundwaters and tidal marsh pore waters, yet they did so with only limited subsurface hydrological data. The controls of groundwater transport from hillslopes to marshes and the effects that marshes have upon residence times and chemistry of discharging groundwater before export to estuaries remain unknown. We have initiated a study in a tidal freshwater marsh in the coastal plain of Virginia to address in detail some of the hydrological and chemical interactions that occur between upland aquifers and tidal marsh soils.

Tidal freshwater marshes occur in coastal riverine areas within the realm of tidal influence but just upstream of appreciable saline influence (Odum et al., 1984). In the lower Chesapeake Bay, tidal freshwater marshes experience a spring and neap tidal range of 80 cm and 100 cm respectively. The marsh vegetation is predominantly Arrow Arum (Peltandra virginica), a broad-leaved perrenial wetland plant that can withstand a wide range of flooding frequency and depth. The organic soils (ash free dry weight percentages ranging between 20 and 40%) contain large amounts of silt and clay deposited from riverine sources onto the marsh surface by flooding tides.

Hydrologically, tidal freshwater marsh soils can be classified as shallow, unconfined aquifers since they are mostly saturated and they possess hydraulic conductivities similar to a fine sand. Tidal flooding of the surface of these aquifers causes rapid changes in pore pressure within the soil that adds complexity to the description of subsurface hydraulics.

The objectives of our initial hydrological studies are three-fold:

- determine pathways of upland groundwater and marsh pore water exchange at a site with maximal potential for such interaction.
- validate a 2-D numerical hydrological model to simulate such an exchange.
- use the model in a sensitivity mode to rank by importance the controls of groundwater discharge to marsh soils and effects upon pore water residence times within the marsh.

In this paper we present and interpret hydrological field data from our first season of data collection without the aid of a modeling analysis.

SITE DESCRIPTION AND METHODS

A tidal freshwater marsh was chosen for study within the James River sub-estuary of the lower Chesapeake Bay (Figure 1). The marsh is approximately 300 m wide and possesses an average surface gradient of 0.006. The uplands in this region are exposures of sandy terraces deposited in fluvial-estuarine environments at elevations up to several hundred feet during higher sea level stands. At the back of the marsh study site the hillslope rises at an angle of 45 degrees and levels off at an elevation of 80 feet.

Instrumentation at the site consists of nests of piezometers installed on the base of the hillslope

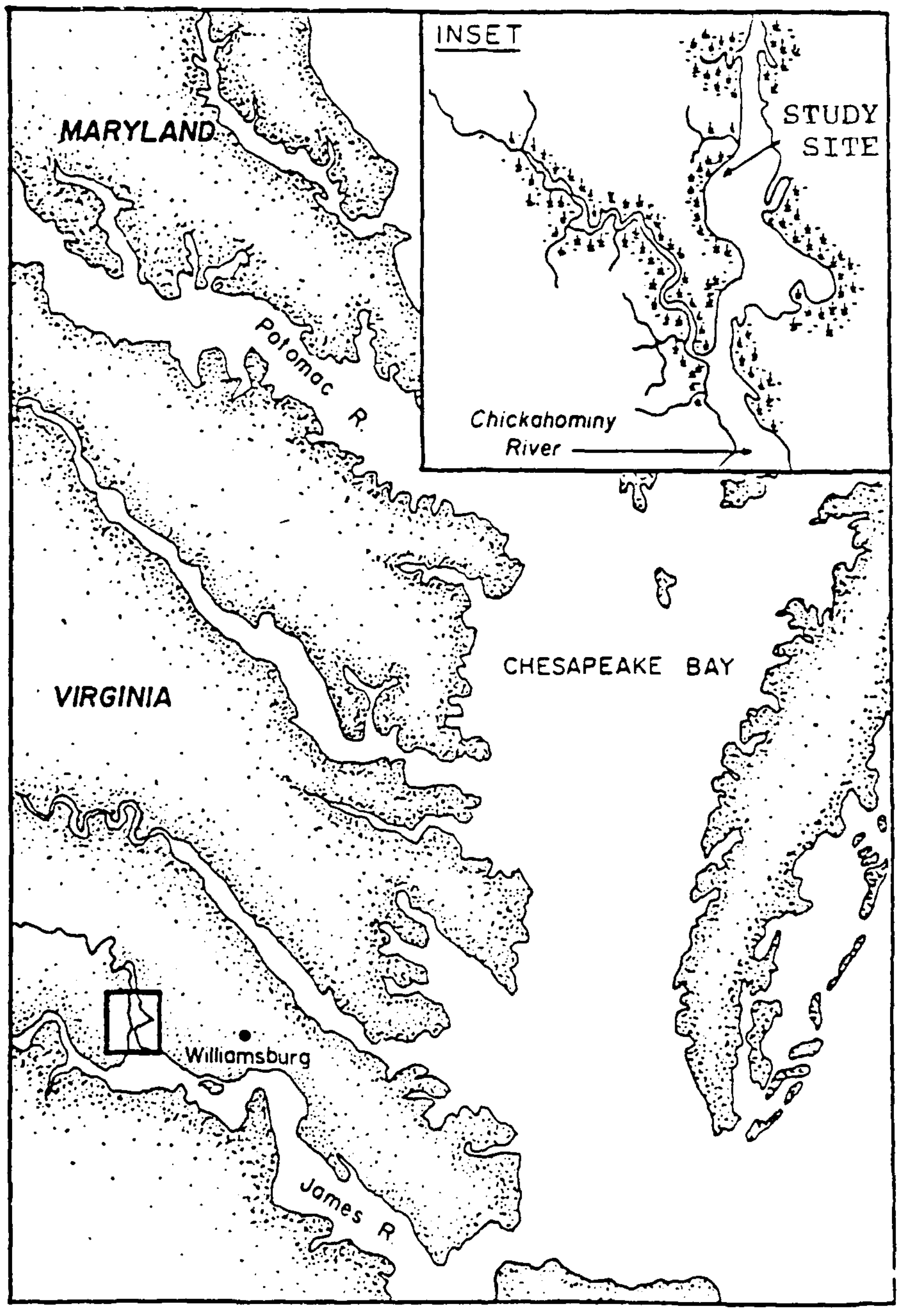

Figure 1. Map showing the location of the study site (37°N,77°W)

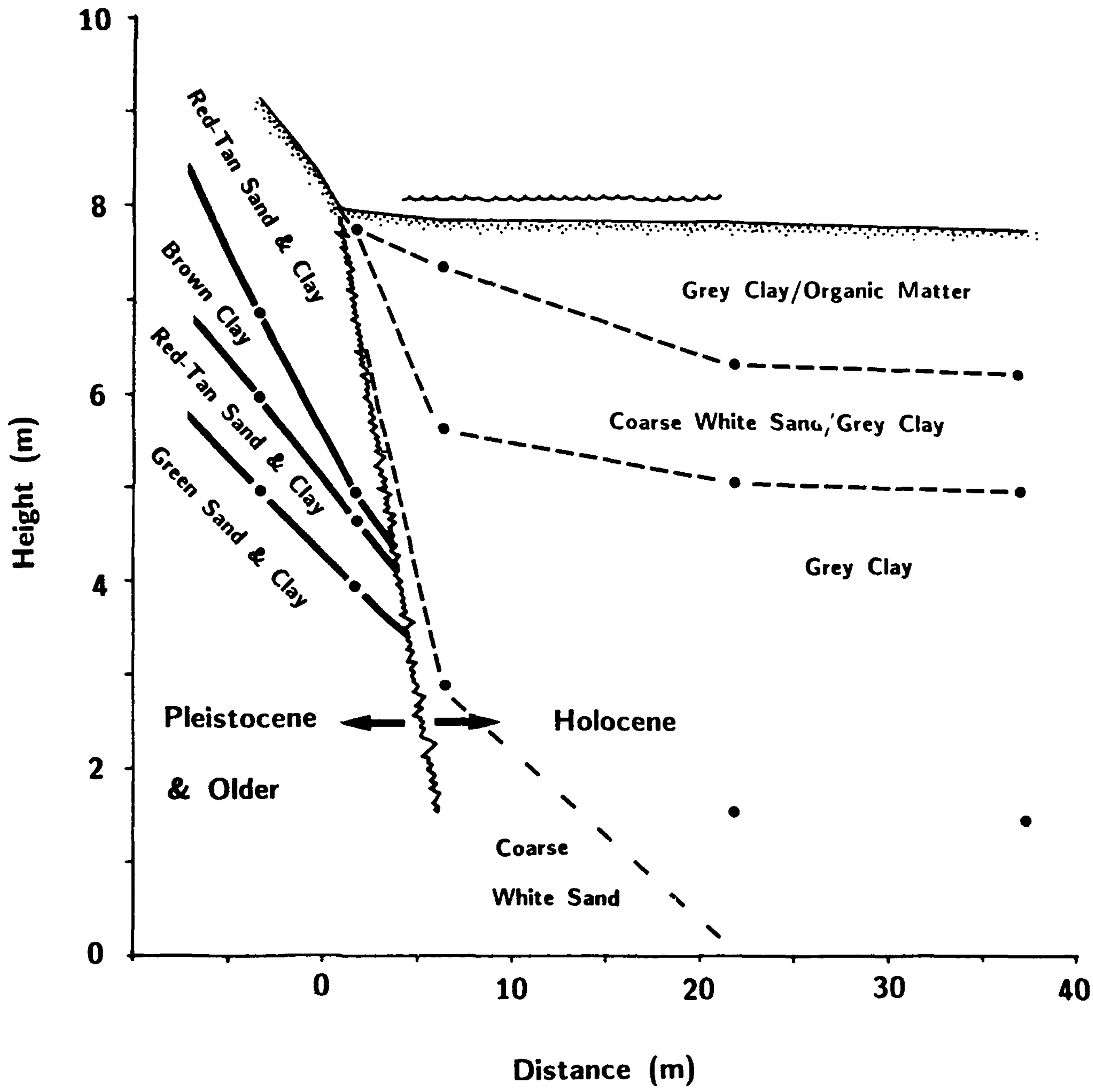

Figure 2. Cross-sectional view of the stratigraphy at the marsh-hillslope interface. Note the presence of an erosional scarp that separates the Holocene marsh deposits from the older soils of the hillslope. The vertical axis is exaggerated five-fold.

and in the marsh on a transect perpendicular to a marsh-hillslope interface. The data base for this paper is drawn from a total of 36 piezometers installed to depths ranging between 0.25 m and 6 m below the soil surface. Small diameter PVC pipe (1/2" i.d.) was used in order to minimize piezometer time lags. Hydraulic head was measured in each instrument with reference to a common datum on daily and seasonal time scales. In addition, measurements of saturated hydraulic conductivity of hillslope and marsh strata were made using the piezometer method and a bail test procedure (Luthin and Kirkham, 1949). Piezometers were also used to remove water samples for measurements of electrical conductance of groundwater.

GEOLOGY

Figure 2 is a cross-sectional view which shows stratigraphy at the study site. An erosional scarp beneath the wetland-upland interface separates soils of different age and structure. Beneath the marsh surface are soils of recent origin and low bulk density, which consist of mixtures of organic matter, clay and sand. Beneath the hillslope are geologically older soils of higher bulk density. Hillslope soils are primarily sands, sandy clays and pure clays. A thin lens of brown clay beneath the hillslope and a mixture of clay, organic matter and sand beneath the marsh have the lowest hydraulic conductivities at the study site (10^{-7}cm/s). The root zone of the marsh soil (0 to 50 cm) possesses the highest hydraulic conductivity (10^{-3} cm/s).

GROUNDWATER DISCHARGE

Figure 3 exhibits the same cross-sectional view of the study site as Figure 2 except that the 5x vertical exaggeration has been removed. The distribution of hydraulic head shown in Figure 3 indicates that a large upward hydraulic gradient exists beneath the hillslope and marsh. This gradient is approximately equal to 1 beneath the marsh hillslope interface and diminishes exponentially as a function of distance into the marsh. Upward hydraulic gradients are greatest in zones of lowest hydraulic conductivity, which indicates that groundwater discharge to the marsh soil from the hillslope may be limited substantially by the low permeabilities of the clay layer on the hillslope and the clay, sand and organic mixture in the marsh.

Note also in Figure 3 that flow lines curve horizontally once the relatively porous root zone is encountered. Vertical gradients in hydraulic head were not detected in the top 2 m of soil in the marsh. Groundwater that is discharged to the marsh soil becomes a component of the pore water budget in the marsh rather than being discharged directly to surface waters. Preliminary calculations indicate that lateral pore water movement in the marsh soil is more rapid than groundwater input; horizontal fluxes are on the same order of magnitude but perpendicular in direction to evapotranspiration fluxes.

Profiles of electrical conductance made to a depth of 2 m in the marsh soil are shown in Figure 4. Note the increase in electrical conductivity corresponds to increases in electrical conductivity of river water during mid-summer. These data implicate the river as the dominant source of water input to the marsh soil. We have continued this monitoring for over one year and have noted a persistent pattern of lower electrical conductivity at 30 cm which we interpret as evidence of shallow groundwater flow from the hillslope.

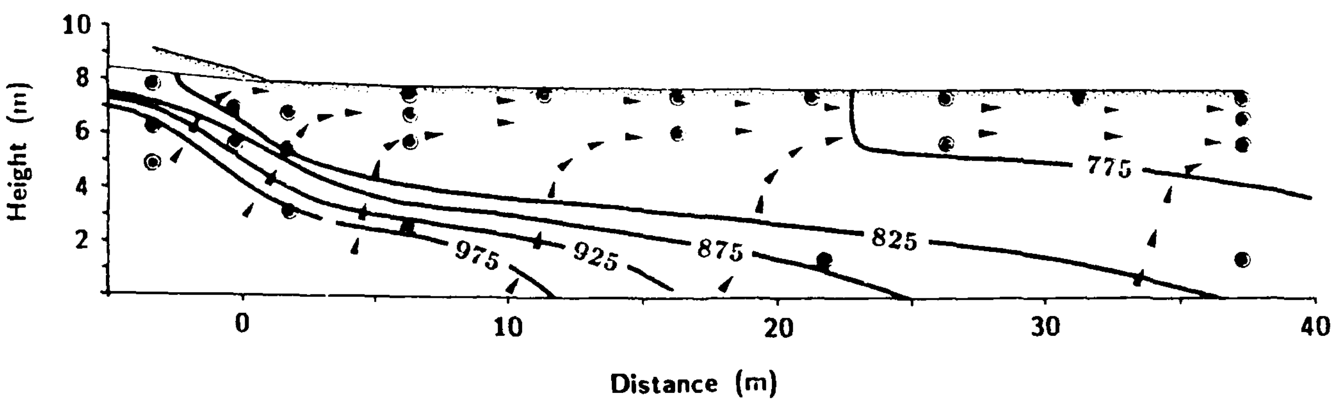

Figure 3. Hydraulic head distribution and subsurface flow pathways between hillslope and marsh. Solid circles are sampling points for hydraulic head (piezometer screens) solid lines are hand contoured equipotentials and arrows indicate flow paths.

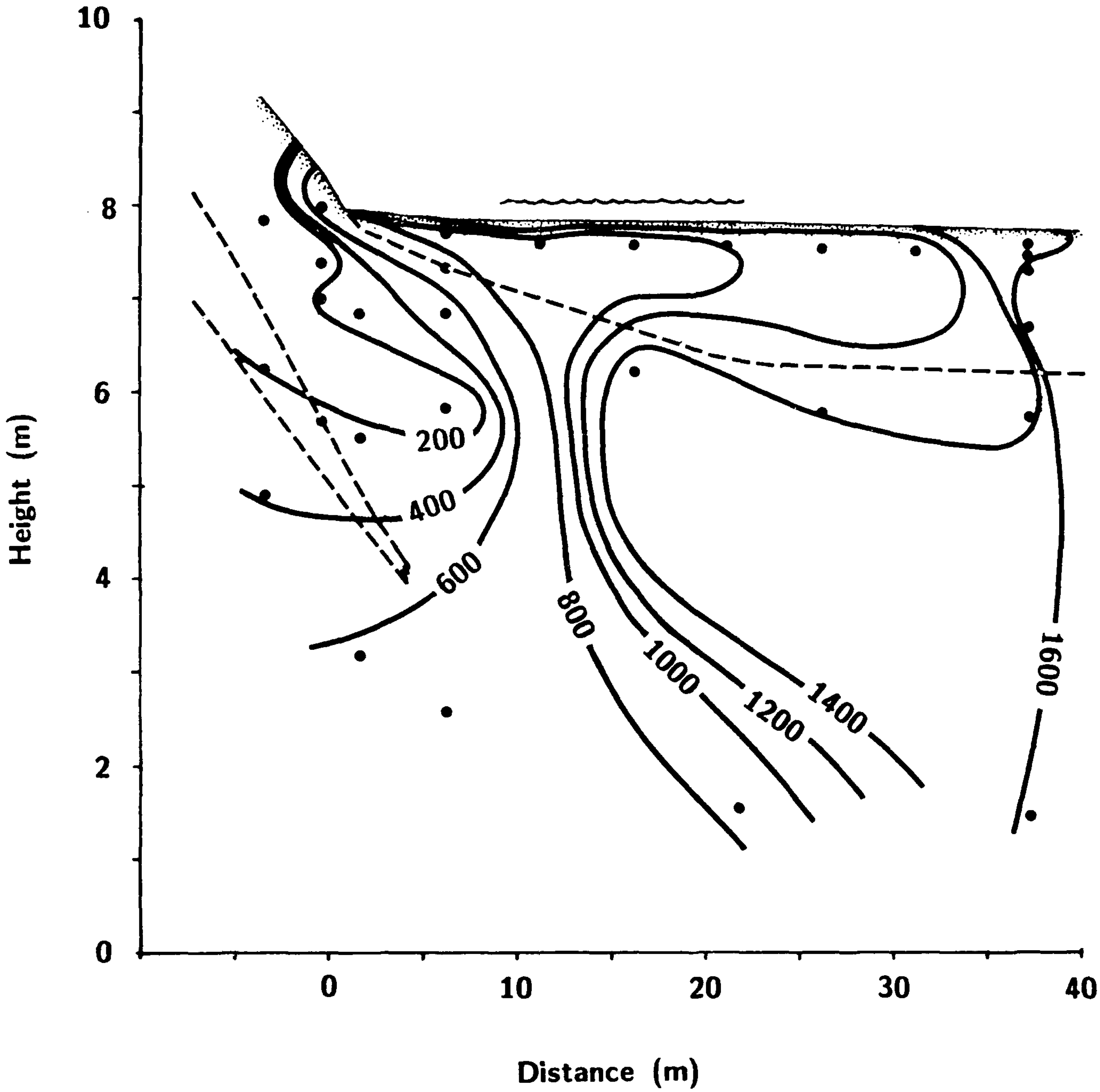

Figure 4. Electrical conductances of groundwater at the marsh-hillslope interface. Solid circles are sampling points and solid lines are hand contoured lines of equal electrical conductance in units of uS/cm. Dashed lines outline the layer of dense clay on the hillslope and the base of the recent marsh soil deposit. The vertical axis is exaggerated five-fold.

DISCUSSION

In 1980 David Lee published data in the hydrological literature on the only flow net we are aware of which shows the influence of groundwater discharge from an upland aquifer on tidal marshes. The flow net is based on sparse data, particularly in the marsh. Lee hypothesized that discharge occurred upward to the surface of the marsh soil, suggesting that groundwater exfiltrates directly into tidal waters on the marsh surface.

Our data have shown that the interaction of groundwater from hillslopes and pore water from adjacent tidal marsh soils is small. Groundwater input is a small component (<10%) of the subsurface water budget of one tidal marsh. Strata of low hydraulic conductivity restrict discharge to the marsh and the transition to a layer of higher hydraulic conductivity at the marsh surface causes flow lines to curve horizontally, effectively preventing direct seepage of groundwater into tidal surface waters. Hillslope groundwater that enters the back of the marsh is subject to solute concentrating effects of evapotranspiration, and the mixing effects induced by infiltration of tidal water while it is transported laterally toward the river under the influence of the slight topographic gradient.

These preliminary data suggest that pore water of our tidal marsh is mostly decoupled from interactions with intermediate or regional scale groundwater transport, even though a major topographic break in slope should have created maximal potential for interaction with deeper groundwater transport systems. Groundwater from the hillslope that originated in the vicinity of the marsh was discharged more effectively into the marsh because transport took place above the low hydraulic conductivity zone at 2 m depth. Thus the presence of a tidal marsh at our study site lengthened residence times of groundwater discharging from the hillslope and altered its chemistry. The effect of land use practices which degrade groundwater quality on hillslopes adjacent to tidal marshes might therefore be ameliorated by soil processes in the marsh such as physical sorption, plant uptake, and denitrification. However, more research is needed at our site and at other marsh sites before the complete physical and chemical role of tidal marshes in modifying groundwater discharge to estuaries is understood.

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chapter nine

Hydrology and Restoration/Creation

Artificially Created Wetlands: Myth or Mysticism

Robert J. Pierce
U.S. Army Corps of Engineers
Washington, DC

Mary C. Landin and Hollis H. Allen
U.S. Army Corps of Engineers
Waterways Experiment Station, Vicksburg, Mississippi

INTRODUCTION

The terms "artificial wetlands" and "marsh creation" conjure up visions to some of synthetic plants, and to others, attribute divine powers to those engaged in such endeavors. We in the Corps submit that "constructed" wetlands are no more artificial than is the water formed by energizing hydrogen and oxygen in a reaction vessel. When the proper elements are combined in a well thought-out design, the outcome is predictable: perhaps not as exact as combining hydrogen and oxygen, but nonetheless predictable. Man's continuing endeavors to assist natural wetland development processes should properly be termed "wetland construction."

Yesterday, we heard the recurring theme over and over that, "Yes, we can construct wetlands successfully" (e.g., Shisler, Lewis and Garbisch) in both fresh and salt water environments. Ed Garbisch even ranked ease of wetland construction in order of increasing difficulty:

1. Fresh tidal marshes;
2. Salt tidal marshes;
3. Fresh, non-tidal marshes with constant water supply;
4. Seasonally flooded floodplains or wet meadows;
5. Temporarily flooded herbaceous marshes (e.g., prairie potholes); and
6. Fresh marshes with groundwater as the only hydrologic component.

Without fail, each of the speakers in the past day and one-half has stated that the most important factor in determining the success of wetland projects is accurately constructing the elevational grade in proper relation to existing hydrology. All of the Corps research confirms that conclusion.

Another recurring theme which addresses the level of success has been the importance of the contribution of open water and fringe upland. Joe Shisler suggested that a constructed wetland should be 50 percent open water and fringed by a protected upland in order to most closely parallel the functions of a naturally developed wetland and to control nuisance species. Robin Lewis and Robert Brooks both observed that seldom is credit given for open water or upland fringe when regulatory and resource agencies are discussing mitigation. The Corps as a regulatory agency and those resource agencies which advise it need to give thoughtful consideration to these areas in formulating mitigation plans.

Are constructed wetlands "identical" to naturally occurring ones? No, but then neither are *two natural wetlands* identical. Furthermore, why should they be? One of the features that makes wetlands so interesting and so valuable is the variability within and between types.

Does each constructed wetland serve all the functions that Paul Adamus attributed to a natural one? No, but then neither do few, if any, natural wetlands. The construction of wetlands is the perfect opportunity to select for those functions most needed in a particular area.

Do soils in recently constructed wetlands demonstrate definite hydric characteristics? No, but then neither do the soils of wetlands behind recently constructed beaver dams or along dynamic meandering streams.

Can we immediately reconstruct a forested wetland? No, but then does clear-cutting the timber on a natural wetland eliminate its value? In many cases the regression to an earlier successional sere increases the diversity and ultimate ecological value of the total system.

Do we know everything about constructing wetlands? Obviously not. But we do have enough technical and applied knowledge to construct wetlands, both fresh water and salt water, under many different circumstances and, in fact, in many situations where natural processes alone would be unable to do the same.

When we were invited to participate in this symposium, we were asked to address three questions:

1. Do we agree with the research needs and strategies for the EPA Wetland Research Program;
2. What research does the Corps have planned or underway; and
3. Will we cooperate with EPA and will our research duplicate EPA's?

We will address each of these questions in order.

EPA RESEARCH NEEDS AND STRATEGIES

Eric Preston began this session with a synopsis of EPA's original wetland research plan. That three-phase plan considered cumulative impacts, water quality, and mitigation as the three most crucial areas of wetlands research to be considered by EPA. Within the mitigation aspect, Eric identified three major thrusts:

1. Evaluate trends and success of past restoration/construction projects;
2. Develop monitoring methods; and
3. Develop design guidelines.

Those three areas have a definite "technocratic" twist to them, i.e, each is designed to provide a directly applicable, policy-oriented tool for use by those in the regulatory program. On the surface, they appear to be relatively straight-forward tasks.

However, after listening to Milt Weller's presentation this morning we have concluded that EPA remains a long way from narrowing and refining their objectives. The compilation which he presented is an impressive array of research topics. Most of them, however, are basic rather than applied. We believe that it is imperative that EPA decide whether it will pursue a basic research program or a more applied one, as outlined in its original charter. Until it has refined its plan, neither the Corps nor anyone else can agree or disagree with the needs and strategies of the program.

Before leaving this topic, let me react to the three approaches discussed by Dr. Weller: functional, comparative, and experimental. It is logical to compare constructed wetlands to those derived from natural processes. We believe, however, that most attempts at a statistical comparison will probably not yield satisfactory results. As discussed earlier in this paper and by many of the other authors of this proceedings, natural wetlands are extremely variable. That natural variation, not only between types of wetlands but between wetlands of a single type, will mean that any type of statistical analysis will require extensive and probably prohibitively expensive replication of samples before any confidence in the results can be demonstrated. We have already encountered this in comparisons of manmade to nearby natural wetlands. Each reference marsh was unique: each differed in age and other critical criteria (e.g. relationship to shoreline, location in a river or coastal system) when compared to the constructed marsh.

Experimental research facilities such as those of the Delta Research Station and the Des Plaines Project will undoubtedly provide very useful data. However, the range of variables and the types of wetlands that can be analyzed using this approach are limited. In addition, since such facilities are themselves constructed, the problem of comparability to natural systems will still remain a question.

The final approach, that of function, is the one that we are pursuing and believe will pay the greatest dividends in the EPA program as well. Functions can be isolated relatively easily, at least conceptually, and for many, there exist well developed methods of analysis. In addition, specific functions can be assessed either on a qualitative basis (e.g., with the Adamus method) or quantitatively for some functions (e.g., habitat evaluation procedure). Most importantly, mitigation can be designed to replace those lost functions which are most important to the overall regional or local ecosystem.

The final concept that needs to be incorporated into wetland mitigation policies and procedures is one first mentioned by Joe Shisler and reiterated by Robin Lewis and Robert Brooks. We cannot construct wetlands in a vacuum. They must be an integral part of the ecosystem. This means that open water and a buffer fringe of upland are as important to the design as the vegetated wetland itself. From an administrative standpoint, we should not expect the private landowner to donate these areas above and beyond the amount of constructed, vegetated wetland that is expected of him. Rather, the overall amount of area needed to mitigate for a particular loss needs to encompass the open-water and upland buffer. This in turn may mean that less wetland vegetation will be planted but abetter system constructed to satisfy the mitigation requirements.

CORPS WETLAND RESEARCH PROGRAMS

The Corps of Engineers has had active research programs on various aspects of wetlands for more than a dozen years. Our original research work was concerned with constructing wetlands from dredged materials, determining the ability of wetlands to rebound from physical stress, and our responsibility to control nuisance aquatic plants. Our research has expanded

greatly over the years. Today, we have over 30 full-time professional wetland scientists as well as hundreds of other researchers in related fields available to interact with us on all of our wetland programs.

Over the years we have constructed wetlands across the country in both fresh water and marine environments. Since recent skepticism has arisen over construction in freshwater environments, I will limit my discussion to examples of some of our efforts there. All of these examples have been monitored for many years by our Waterways Experiment Station (WES) (Newling and Landin, 1985).

In 1973, we built a fresh-water marsh at Miller Sands Island: a 235-acre site in the Columbia River in Oregon. The original 12-acre wetland has not only survived and flourished, but has increased three-fold in size over the past 14 years. This marsh was planted with slough sedge, Lyngbye's sedge, arrowgrass, broadleaf arrowhead, tufted hairgrass, and several other "minor marsh" species. It was also colonized by other species, and is now a very diverse intertidal marsh. The marsh is heavily used by nutria, muskrat, waterfowl, and waterbirds, while the island itself is frequented by the endangered Columbia White-tailed Deer and 108 species of birds, including numerous bald eagles who fish the channels around the marsh. This island is part of the Lewis and Clark National Wildlife Refuge. The three habitat development areas (upland, wetland, and dune) on Miller Sands have been equal or greater in value for wildlife, fisheries, benthos, and vegetation biomass and diversity when compared to three nearby natural marsh/uplanded sites (Landin, et al. 1987).

The Pointe Mouillee wetland is a 4600-acre site built and restored by the Corps Detroit District and the Michigan Department of Natural Resources in western Lake Erie. Rather than planting it, the site was allowed to colonize naturally and sediment has been allowed to accumulate in backwater, protected areas to raise the lake bottom and encourage the growth of emergent species. The key to success at Point Mouillee has been achieving the correct engineering grades and elevational goals. This site is part of a state wildlife management area managed by the State of Michigan. It receives tremendous waterfowl, shorebird, and waterbird use during migration, and waterbirds nest in the wetland and on the dikes. The long-term management strategy for the site includes establishment of a visitors' center, hiking trails, fishing piers, a marina, and other natural resource and recreational activities. Pre-disposal site concerns about possible contaminants in the sediments affecting wildlife and fisheries turned out to be unfounded, and contaminants have not been a problem. While full achievement of anticipated emergent marsh goals at Pointe Mouillee has been slowed considerably by e record high water levels in Lake Erie, we fully expect those goals to be met as lake levels drop (Landin, 1984).

Lake of the Woods is a Corps-built, freshwater wetland, at Warroad, Minnesota, on the border with Canada. This site was formed by required maintenance dredging of the Warroad River and harbor. It was colonized by river bulrush and cattails. Both the portions of the island that are emergent and those that are bare are heavily used by black terns and other waterbirds in the summer months. This site is a cooperative venture, built and monitored by the Corps St. Paul District, with a large share of the monitoring being handled by the Minnesota Department of Natural Resources. This site has also been temporarily affected by record high water levels, but in recent months water levels have dropped and the island wetland is continuing to develop (Landin, 1985).

As with any research effort, we are not without our share of early failures. One of the earliest Corps marsh construction efforts was at Windmill Point in the James River, Virginia. There, our Norfolk District constructed a rectangular berm and filled it to design elevation with dredged material from the adjacent channel. We had contracted with Ed Garbisch to plant the site, but before he and his crews could do so, the area colonized itself, naturally. Ed was left with only the berms to plant with herbaceous vegetation. The marsh was exceedingly productive between 1975 and 1983, then, much of it was washed out due to severe river flooding. That effort did teach us, however, that islands shaped to match river hydrology are much more effective at withstanding floodwater erosion than rectangular ones, and that woody vegetation planted on the dikes probably would have held the island against river flooding (Landin and Newling, 1987).

Much of our recent research has dealt with the critical issue of controlling erosive energy. Marsh development technology has expanded to the point where it is now possible to develop marshes in areas that are exposed to moderate to high wave energies. These marshes tend to be those directly exposed to average fetches of more than 5 miles. Examples of such efforts, developed and employed by the WES, include using expedient breakwaters and new techniques of stabilizing plant stems (Allen, et al., 1986).

One method of establishing marshes in a moderate to high wave-energy environment is to couple breakwaters and transplanted sprigs landward of the breakwater. Allen, et al. (1986) reported successful use of a floating tire breakwater to develop a marsh in Mobile Bay to stabilize a dredged material island. Recently, similar breakwaters have been used by WES and

the Corps St. Louis District to develop marsh and transitional wetland vegetation for shoreline protection at Carlyle Lake, Illinois.

Other effective methods to establish marshes in moderate to high wave-energy environments exist that are more visually attractive and possibly less expensive than breakwaters. The concept is to strengthen the attachment of the plant to the substrate to reduce the likelihood of its being washed out by wave attack and thereby avoid the necessity of a breakwater.

One such method is a plant roll. A plant roll is constructed by placing soil and transplant clumps of a marsh species such as smooth cordgrass or softstem bulrush on a strip of burlap. The sides and ends of the burlap are brought together around the plants and fastened with metal rings. This creates a long roll of plants and soil that is then air jetted into the substrate. Such a method has been used successfully to protect dredged material shorelines in Mississippi Sound and Mobile Bay, Alabama. Like breakwaters, the plant roll technique has also been used recently to establish a softstem bulrush marsh at Carlyle Lake, Illinois.

One other promising method for use in higher wave-energy environments is an erosion control mat laid like carpet on a substrate. After placement, single stems of marsh grass are planted in the mat through slits cut in the material. This technique has been used successfully in demonstration plots along the Gulf Coast at Bolivar Peninsula, Texas, where the marsh grass has been exposed to fetches of 22 miles.

Day-by-day, the Corps and other individuals and groups actively constructing wetlands are advancing the state of our knowledge. The basics are well understood (U.S. Army Corps of Engineers, 1986). In fact, the Corps has been teaching a practical wetland construction methods course for the past eight years. By the end of this year, we will have available for distribution a video cassette on shoreline stabilization using coastal wetland vegetation.

I don't want to leave you with the idea that there aren't problems, especially in using construction as a form of mitigation in the regulatory arena. By and large, however, these problems are not technical. Rather, they result either from imprecise definition of the objectives of the mitigation and methods to accomplish them or from insufficient compliance monitoring. Both problems are symptoms of a more pervasive problem - the lack of funds and personnel to administer the program effectively.

Last year Congress cut our budget and prohibited us from transferring additional funds into the program. This year we have gone as far as to have President Reagan formally sign a proposed amendment to the budget, specifically allowing us to transfer $5 million more into the budget; not to make us fat, but just to keep our regulatory program afloat. Despite the resource constraints, however, we are not standing still in this arena. A year ago our Baltimore District completed a study of compliance and effectiveness of mitigation efforts associated with permit projects. While initial compliance and success was only about 50 percent, after prodding of permittees, success is now seen in 15 of 16 projects. The Corps Norfolk District found similar results in a study they conducted. Approximately one-third of the projects they reviewed were deficient and were required to undertake corrective measures. Similar studies are underway in the New England and Jacksonville Districts.

Because of the workshop theme, we have concentrated our presentation thus far on wetland construction. At this point we will diverge briefly and list some of the other research areas we are currently pursuing.

This past February we began a one-year test of our multiple parameter approach to delineating wetlands. This test culminates four years of intensive research, field verification, review, and revision. The response to the effort has been very positive, and we look to this coming year for a unified method endorsed by both the Corps and the EPA.

We are continuing our studies of the last five years on contaminant availability from sediments by both marsh and upland plants and animals. We have been coordinating this effort with American scientists and have established an International Advisory Board with whom we have been working closely. In addition to short-term studies, we will be monitoring over the long-term the movement and fate of contaminants from freshwater sediments at Tynes Beach, Buffalo, New York, and marine sediments at Black Rock Harbor, Connecticut.

Our aquatic weed control research program has been in operation for over 20 years. In that period, we have evaluated and developed biological, chemical, mechanical, and physical methods for controlling nuisance vegetation. One aspect of this program has been to seek out, import, quarantine, and evaluate for effects on indigenous communities those natural control organisms of exotic plant species which have been introduced into the United States.

Our major wetlands research thrust for the next several years centers around the various functions of wetlands and how to place a value on them.

Over the last two years, working closely with the Federal Highway Administration, we have computerized the evaluation system developed by

Adamus (1983) and updated the literature compilation (Adamus and Stockwell, 1983) upon which it was based. This effort has yielded a user-friendly program (Wetland Evaluation Technique or WET) for IBM PC compatible computers which is already available, as well as a revised literature compilation which will be available in the next four months.

Because WET is based on the technical literature many data-gaps exist. We have prioritized data needs (Clairain, et. al., 1985) and intend to fill those sequentially as funding allows. As our top priority, we are beginning the second year of a 3-year, detailed study of the biological, chemical, and physical functions of a large tract of bottomland hardwood wetland on the Cache River in Arkansas. From the bottomland hardwoods, we will, in time, be moving to the wetlands of the north central region, the southwest, and on through our priority list.

COOPERATION AND DUPLICATION

We have briefly summarized our research activities in wetlands and, in conclusion, we will comment on the final question that we were asked to address: will we cooperate with EPA and will our research duplicate its research.

Throughout the history of our wetland research programs we have maintained close liaison with those interested in the subject. In recent years and thanks in large part to the organizational efforts of Hank Sather and Bill Wilen, we have actively participated with the Interagency Wetland Value Assessment Group. We do not have, nor can we foresee, any reason toterminate that cooperation.

Finally, because of the undefined nature of EPA's program, it is not possible to comment on how duplicative it will be. From a philosophical standpoint, however, we must close by posing the more appropriate question: "EPA, as the new kid on the block, will your study duplicate our ongoing programs?"

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Hydrology and Hydraulic Requirements of Successful Wetlands

Harold A. Vance
John M. Tettemer & Associates, Ltd.

WHAT IS A SUCCESSFUL WETLANDS PROJECT?

A successful wetlands project is one that consistently and reliably provides food and shelter for birds and animals whose existence may be otherwise threatened. With up to 90 percent of the nation's historical wetlands removed for agriculture and urbanization, the remaining wetlands and the ones we create are under tremendous pressure. We go to great lengths to preserve the last remaining California condors, sea otters, and giant pandas. What would we have done 60 million years ago when it was becoming too cold for the dinosaurs? I believe we would have turned on the heat. If we are going to save these species, we have to manipulate their environment. If we are going to have successful wetlands, we have to create and maintain a hydrologic and hydraulic environment that is supportive of the birds and animals. It must be buffered not only from the impacts of man, but from the extremes of nature. It will not be natural. The unreliability of nature is a luxury we cannot afford.

We can summarize the requirements of a successful wetlands project as follows:

1. Clear objectives
2. Definition of hydrologic and hydraulic micro-system (within the wetlands) requirements
3. Account for benefits and threats of macro-system (surrounding the wetlands)
4. Ability to manage and control macro-system
5. Ability to manage and control micro-system
6. Management and financing structure

OBJECTIVES: CLEAR AND REALISTIC

Success depends on clear and realistic objectives. They need to be defined first in biologic terms--what kinds of birds and animals are to be supported, how many, and what times of the year. With these objectives in mind, the biologic requirements can be established: What kinds of vegetation; what configuration of vegetated areas; what relationship between land, vegetation, and water; how much area; how much open water; how much vegetation; how high the land; how deep the water; and what requirements are there for controlling and directing water within the wetlands. These biologic requirements allow the hydrologist to determine the hydrologic and hydraulic requirements: average annual water requirements for evapotranspiration, flushing, and possible losses; minimum and maximums that can be tolerated; control structures such as levees, gates, weirs, and valves; and the necessary water surface elevations to achieve the desired performance. These requirements are imposed on the site to determine feasibility and cost. The biologist, owner, and engineer work together to develop objectives that are realistic in terms of site constraints and financial limitations.

UNDERSTANDING THE MACRO-SYSTEM

The hydrology and hydraulics of the wetlands project are influenced by those of the macro-system, both positively and adversely. Positive influences include possible water supply from surface flows and groundwater. These can be made up of storm runoff, wastewater treatment plant effluent, and agricultural or urban irrigation tailwater. Adverse influences include interruptions and fluctuations in the water supply due to upstream diversions, floods, drought, and changes in groundwater levels; erosion; sedimentation; and water quality changes.

Wetland projects should be designed to take full advantage of positive attributes of the macro-system. This includes site selection for the wetlands project, which may be a degraded wetland that flourished in the past. The creation and enhancement of wetlands is an expensive undertaking, so sites which have low value for other uses, which have access to a water supply, and which require a minimum of grading and control structures are preferable over alternatives which do not.

The following macro-system features can influence the success of a wetlands project:

- Upstream reservoir to regulate flows and provide a reliable water supply

- Upstream wastewater treatment plant to provide or augment water supply
- Upstream urbanization to provide year-round landscape irrigation runoff
- Possible adverse water quality due to urban, agricultural, or industrial runoff
- Siting of wetlands: in stream or off-stream
- Potential for flooding
- Potential for erosion of wetlands or diversion system
- Potential for sedimentation of wetlands
- Potential for reduced water supply due to drought or upstream diversion
- Groundwater regime: stable, declining, or fluctuating
- Availability of groundwater to provide or augment water supply
- Alternative water supply during periods of low flow or unacceptable quality
- Development policies in watershed; flood peak mitigation, erosion control, concrete channels, greenbelts
- Downstream water rights holders and water quality interests.

Just as a project should take full advantage of macro-system features which reduce costs and improve reliability, so should a project be buffered from macro-system features injurious to its performance. An unreliable or fluctuating water supply should be supplemented. An eroding stream should be stabilized. Sediment basins should be provided to trap sediment before it enters the wetlands. Water quality upsets should be monitored and a supplemental water supply utilized until acceptable quality is restored. Watershed development policies aimed at mitigating the impact of development on flood peaks, erosion, and sedimentation should be supported.

CONTROLLING THE MACRO-SYSTEM

Optimum productivity within a wetland is achieved when the land, water, and vegetative resources are maintained in the proper balance for production of food and shelter. Operation of the wetlands may resemble running a farm. The wildlife and waterfowl depend on the wetlands, and we invest time and money in preserving the land for wetlands; therefore, it is logical that we manage wetlands to maximize their productivity. The hydraulic requirements of wetlands management may include:

- Providing flow to maintain desired water depths and satisfy evapotranspiration--this may involve a stream diversion, well, or pipeline for delivery of fresh or reclaimed water
- Providing flushing flows
- Ability to drain and flood basins for seed production, vector control, maintenance, etc. This may involve delivery and drainage canals, gates, and pumps
- Ability to withstand flooding, erosion, and sedimentation. This may require levees, revetments, stabilizers, and sediment basins
- Measurement of flows
- Monitoring of water quality
- Changing from one source of water to another.

PERMANENT MANAGEMENT

Today's engineered, high-productivity wetlands need ongoing attention, just like agricultural operations. A source of funding must be established, such as an endowment by the sponsor, a wetlands district, or a cooperative agreement among participants. A qualified operating entity should be chosen to provide ongoing management. The funding mechanism and the operating entity should be established at the preliminary planning stage because the objectives of the project can be realistically established only when the level of funding and operating entity have been defined.

The approach outlined above is being used on several successful projects, three of which are described below.

Dove Canyon Country Club

The Dove Canyon Country Club project in Orange County, California, required the removal of somewhat less than 10 acres of wetland and riparian habitat. To compensate for the removal, a mitigation and enhancement plan is being implemented which will include:

- Water conservation through the use of reclaimed water. The existing wastewater treatment plan will be expanded
- A new lake which will provide waterfowl and riparian habitat, flood control, sediment control, reclaimed irrigation water storage, and aesthetic benefits
- A perennial stream in a former dry canyon
- Three new wetlands ponds on the National Audubon Society Starr Ranch Bird

Sanctuary

- Creation of a new riparian forest along the fringe of the lake and on islands or peninsulas to be created within the lake by filling
- A monitoring system for water quality leaving the project and entering the National Audubon Society Sanctuary
- A pump to recycle unacceptable water back through the tertiary treatment facility.

The planning of this project took place with input from many sources. The U.S. Fish and Wildlife Service, California Department of Fish and Game, California Division of Safety of Dams, California Division of Water Rights, National Audubon Society, U.S. Army Corps of Engineers, Santa Ana Mountains County Water District, and County of Orange were fully involved along with the project development and hydraulic engineers and the owner. The resulting mitigation and enhancement plan will not only replace existing willow and sycamore woodlands, but will upgrade mulefat scrub areas to willow thickets and add permanent open water and wetlands ponds for greater productivity and diversity. The facilities will be under professional management to assure long-term viability. The wetlands to be created on the National Audubon Society sanctuary will be operated by the Society.

San Joaquin Marsh Wetlands Enhancement Project

This wetlands restoration and enhancement project is a cooperative venture of the public and private sectors sponsored by the Irvine Company in Irvine, California. It involves the enlargement and enhancement of an existing degraded wetlands and the establishment of a permanent professional management structure. It provides a good example of regional habitat master planning and demonstrates that a team of qualified professionals can produce good results. The participants in the development team are the U.S. Fish and Wildlife Service, California Department of Fish and Game, University of California Reserve System, the City of Irvine, and the Irvine Company.

The objectives set for the project included creation of additional open water by removing cattail mats, providing drainage facilities, providing a supplemental water supply, providing interbasin controls, providing additional riparian woodlands, and providing clapper rail habitat.

The project site is adjacent to the San Diego Creek flood control channel. The water supply alternatives considered for the project included a stream diversion from the channel with a sediment control basin. However, it was determined that low flows in the channel contained urban and agricultural pollutants that were unsuitable for the wetlands; therefore, the supplemental water supply will be provided by pumping existing wells and drilling one new well, rather than by building the diversion.

Interbasin controls will be provided by installing additional levees to provide separation between basins, and by the installation of control gates located to provide maximum operational flexibility.

A new drainage outlet to San Diego Creek will be provided, with a pump for draining the lowest basins. A drainage canal will be built through the wetlands to provide each basin with access to the drain.

New clapper rail habitat will be provided by excavating a series of channels that will provide maximum edge between cattails and open water. The channels are spaced to accomodate the reach of mosquito control equipment.

Willows and sycamores are being planted to provide additional woodlands. Features that were evaluated and determined not cost effective were an additional lake and the remodeling of existing duckponds. Design of the tree planting areas and clapper rail habitat underwent several changes as the team members evaluated the advantages and disadvantages of various alternatives.

Upper Newport Bay

Upper Newport Bay is the drainage outlet for San Diego Creek in Orange County, California. Prior to settlement of the area, the creek did not extend to the bay, but terminated in a large marsh which merged into the bay. The area was an outstanding example of freshwater and salt water marsh, supporting large flocks of resident and migratory waterfowl. Urbanization of the drainage area led to the excavation of San Diego Creek Channel. A flood in 1969 caused large volumes of sediment from the drainage area to be washed into the channel and into the bay. The upper bay filled with sediment. The marsh was converted to mudflat and upland. This is an example of the overriding influence of the macro-system on the micro-system.

For several years government agencies, elected officials, the press, environmental interests, and community leaders searched for an understanding of the problem and an approach to restoring the bay. It was concluded that Section 208 of the Clean Water Act offered a mechanism. A study was undertaken under the 208 program to evaluate alternatives and recommend a long-term plan to restore and maintain the bay. The hydrologic questions to be answered were: What range and frequency of flows should be expected in the future, and what are the sources and

amounts of sediment? The hydraulic question was what are the elements of an economically feasible sediment control system to protect the bay in the future?

The 208 study provided the answers. Sediment sources were identified as natural foothills, farmlands, and construction sites. Urban areas produced little sediment. As the area develops, many sources of sediment will be eliminated. For the remaining sources, sediment basins in the San Diego Creek and in the upper bay will provide controlled locations for capturing the sediment for periodic removal.

A local cooperative management group was created to raise funds and implement the restoration and maintenance program. Participants include the State Department of Fish and Game, the County of Orange, the Cities of Newport Beach and Irvine, the Orange County Beaches and Harbors District, and the Irvine Company. Two of the in-channel basins have been built and cleaned out. The bay has been restored and two in-bay basins are in place. The bay configuration was engineered by the biologists and engineers to assure optimum flushing by tidal action, provide marine and marsh habitat areas, and provide efficient sediment traps. The cooperative group acts as an executive management committee to assess its members and otherwise raise funds, develop dredging contracts, monitor performance, and plan necessary maintenance. This project is an outstanding example of local initiative to utilize the principles of hydrology and hydraulics in a political and managerial environment to create an excellent wetland.

SUMMARY AND CONCLUSIONS

These are examples of successful wetlands restoration and creation projects. They have several features in common: clear, realistic objectives; understanding of the hydrology and hydraulics of the macro-system in which they exist; clear definition of the hydraulic requirements of the micro-system; ability to regulate or control both the macro- and micro-systems; and professional management. The projects were developed by multi-disciplinary teams including biologists, regulatory agencies, hydrologists, and land owners/financiers. As the pressure on remaining wetlands increases, the outlook is promising for successful restoration and enhancement projects embodying hydrologic, biologic, and management skills.

Creation of Streams, Wetlands and Riparian Corridors to Improve Environmental Quality Within Developments

Robert J. O'Brien
INTER-FLUVE, Inc.

INTRODUCTION

In many areas of the Rocky Mountain west, development commonly follows river floodplains due to the overall ease of construction and the engineering characteristics of flat valley sediments (see Legget, 1973). Although this development activity is economically attractive, the environmental costs are commonly high, i.e., impacts to fish and wildlife habitat, reduction of open space, and loss of recreational opportunities. Contrary to this trend, this paper will present an overview of a residential housing project that has improved environmental quality while providing a critical link for the extension of an existing greenbelt system.

Greenbelt Plain, Boise, Idaho

In 1982, the City of Boise, Idaho adopted long term goals for the ecologically sensitive riparian land adjacent to the Boise River, an area that had been targeted for growth. The creation of a greenbelt plan was thought to be the best means of protecting this central and popular amenity while enabling the city to expand its park system. The primary goal of the plan was to regulate land use within the floodplain and adjacent lands in a manner consistent with the preservation, protection or enhancement of existing fish and wildlife resources. The plan would also serve to facilitate conveyance of the 100-year flow without loss of life or property damage.

The city began the process with community meetings to determine species and/or areas of special interest. A small group of environmental consultants (Inter-Fluve, Inc., Resource Systems, Inc. and Ecological Design Associates) was then retained to identify important habitat for bald eagles, blue herons, brown and rainbow trout, and a variety of waterfowl. Surveys were conducted to determine overall habitat conditions within the floodplain, the need for preservation, and the potential for habitat enhancement.

The result was the classification of riparian lands within the 100-year floodplain into three categories representative of the major types of habitat present. Class A lands were those wetland complexes that provided good habitat for a variety and/or an abundance of species. All land within the 200 foot setback from the 100-year flow was also considered as Class A land regardless of its condition. No development was to occur in these areas. Class B lands were considered those areas with potential for habitat restoration, and therefore suitable for conditional development. Class C lands were characterized as having poor wildlife habitat, the least potential for habitat improvement and therefore suitable for development.

The Development at River Run

While the greenbelt plan was being formulated, the River Run Development Company was completing a unique low density residential project adjacent to the Boise River. This development was characterized by a design that relied on the aesthetics provided by a natural riparian setting. To further enhance this character, the developer proposed to construct several small streams as water amenities. However, the water rights necessary for this work were opposed by the Idaho Department of Fish and Game on the basis that usable salmonid habitat within the Boise River was close to critical levels at low flow. To resolve the problem, Inter-Fluve, Inc. proposed to design and build a series of viable trout streams within the development. This design met with approval by the Department and the water rights were granted.

The new channels were to create spawning and rearing habitat for brown and rainbow trout. This type of habitat was critically needed along the Boise River, since dam construction and years of flow regulation had reduced the river's capacity to create this habitat. In addition, the streams were integrated with lot dimensions and building setbacks to facilitate a small riparian zone. This was planted with native vegetation to attract various species of wildlife. The project resulted in several miles of fish and wildlife habitat. Following construction, the developer was praised by the local media and named conservationist of the year in Boise.

The Development at Spring Meadows

Due to the success of this work, the developer acquired six more parcels of land along the Boise River. However, under the new greenbelt plan,

some of this land was classified as Class A due to the existence of a mature stand of black cottonwoods and the fact that some of the land was inside the 200 foot setback from the 100-year flow. These trees were located adjacent to a major riffle in the river and provided habitat for blue herons and other wildlife. Additionally, some of these trees were known to provide winter perch sites for bald eagles.

Unfortunately, the majority of the land within these parcels was characterized by relatively poor overall wildlife habitat. This condition resulted from a history of livestock grazing use that eliminated the understory, prevented regeneration of the cottonwood stand (which was considered to be in a decadent phase) and severely reduced vegetative diversity. This grazed riparian forest was very different from a riparian forest located directly across the river. This natural area was characterized by a thick understory and an abundance of wildlife. For example, vegetation included black cottonwood, several willow species, silver maple, wild rose, false indigo, black hawthorne, white alder, currant, box elder, Russian olive and a host of sedges, reeds and grasses. This diversity was reflected in the results of plant density counts, which are shown in Table 1.

Table 1
Plant Density (# of plants/acre)

	Grazed Riparian Forest	Riparian Forest
overstory	60	144
understory	190	1,438

Since the availability of food and cover for wildlife is highly correlated with tree and shrub canopy area (Hays et al., 1981), wildlife use within the grazed riparian forest was considered to be severely restricted.

PLANNING CONSIDERATIONS

In addition to the development guidelines established by the creation of the greenbelt plan, the City of Boise also had specific goals for the parcels comprising the Spring Meadows project. In particular, the City sought to accomplish the following: 1) extend an existing primitive walking path through the property to connect with another path at a large city park south of the property, 2) extend an existing bike path in a similar fashion, 3) provide an additional public access point to the greenbelt, and 4) develop a small public park adjacent to this area.

From a construction standpoint, the ownership of riverfront property is a benefit to a private landowner and a major selling point for a developer. In this respect, the existence of public walkways, a bike path and an access road to the greenbelt would add little to the project and might even be considered a detraction. In addition, although the city land use plan allowed a maximum of 1,049 dwelling units to be built within the project area, the developer was planning to build only 442 units. This lower housing density, combined with improvements to the riparian area, would facilitate an appreciation for the unique resources of the property and retain as much of the natural wildlife values as possible. The developer recognized that the city's goals were potentially beneficial to the community, and decided to help provide these amenities as long as the low density project would remain economically viable.

Since the city did not have the funds to acquire the necessary land for public amenities, the developer proposed to give 5.4 acres to the city to facilitate their construction. In addition, the developer also agreed to build a road to provide access to the greenbelt. In return, the city gave 5 acres to the developer for inclusion into his project. Also, the developer was allowed to encroach somewhat on the 200 foot setback by making this setback coincident with a required 25 foot setback from natural water features. This encroachment was acceptable to the city as long as the developer made some improvements to the greenbelt. To help achieve these goals, several development alternatives and their associated mitigation plans were presented to the city.

DEVELOPMENT ALTERNATIVES

The impacts to wildlife, fisheries, and recreation associated with three major development alternatives for the newly acquired parcels of land were reviewed and presented to the city. The three alternatives were: 1) maintenance of existing land use and conditions, 2) development following the planning and zoning guidelines allowed by the Boise Greenbelt Plan and 3) low density development combined with fish and wildlife habitat improvement.

The first alternative would have facilitated the eventual loss of the cottonwoods and the Class A habitat they provided, as well as contributed to the continued decline of other wildlife habitat. The second alternative involved a variety of land uses directly or indirectly detrimental to the desirable species of wildlife such as: relatively high density housing (the maximum of 1,049 dwelling units), the construction of a warm water lake, and construction of the bicycle path within the flushing distance of herons and eagles. This alternative would have resulted in the migration of existing wildlife to less pressured areas and created thermal pollution within the trout streams

of the River Run project and the Boise River. The third alternative, which involved low density housing and improvement of the riparian zone, was chosen by the city as the best choice to meet their goals for the area.

The conceptual basis for this alternative revolved around maximizing the open space adjacent to the Class A lands and completing a variety of habitat improvements within the 200 foot setback from the 100-year flow. These improvements included plans for the construction of over 1 mile of new trout stream, the creation of an emergent/scrub-shrub wetland as a vegetative barrier to prevent human disturbance within the Class A areas, the revegetation of the area to help restore riparian forest conditions, the relocation and reconstruction of an existing water supply ditch (Loggers Creek), and the establishment of native vegetation in other areas. In addition, roughly 38 acres of the 92 acre total would be set aside as open space in order to provide a critical link for existing greenbelts north and south of the property. Some of the more salient aspects of the design of these features are outlined below.

DESIGN HIGHLIGHTS

Loggers Creek Relocation and New Channel Construction

Several decades ago, Loggers Creek was relocated and channelized by a local landowner to meet the water needs of his ranch. Since that time, the canal had been utilized by a local water company to supply irrigation water for farmers and ranchers in the Boise River valley. A riprap diversion structure and concrete headworks were constructed on the Boise River to create the head necessary to provide a minimum of 45 cubic feet per second for these users. As previously mentioned, the trout streams created for the River Run residential area also utilized this system for their water source. Due to the very low slope of the canal (0.0018%), a sandy stream bed, and steep banks comprised of spoil material, this channel provided virtually no habitat for fish or wildlife.

There were two phases to the design of this project. In the first phase, Loggers Creek was to be returned to its old alignment to utilize the vegetative cover and diversity provided by an existing riparian forest. This alignment also increased the slope by reducing the channel length, and thereby facilitated the creation of habitat for brown and rainbow trout. The lower end of the channel was to connect with the trout streams previously created for the River Run development. Revegetation with understory species was to create a storied riparian zone, and additionally, create a natural barrier to the Class A lands adjacent to the Boise River in this area.

In the second phase, the old headworks were to be removed and Loggers Creek extended upstream for a distance of about 1 mile to create even more trout habitat. In this area, which contained the raptor habitat, the stream bank across from the homesites would grade from an emergent wetland margin to an impenetrable scrub-shrub wetland understory. This understory was also designed to complement the riparian forest adjacent to the Boise River. Although the stream bank next to the homesites would be accessible to homeowners, it would be protected by a 25 foot riparian setback comprised of native vegetation.

The net result was to be the reestablishment of a storied riparian forest and the creation of more than one mile of contiguous fish and wildlife habitat. In addition, the removal of the headworks for old Loggers Creek and its diversion structure would create another riffle in the Boise River, and thereby, improve passage for river rafters, increase angling opportunities, improve brown and rainbow trout habitat and provide additional feeding and nesting habitat for herons, eagles and other raptors.

Emergent and Scrub-Shrub Wetland

The soils of the riparian and wetland areas within the Boise River valley fall into two broad categories: gravelly sandy-silts within the riparian edge and sandy silty-clays within the valley flat. These soils support three different vegetative communities depending on the amount of moisture present: 1) a sedge/spikerush, 2) a palustrine/scrub-shrub and forested wetland composed of varying proportions of cottonwood, birch, willow, hawthorne and alder and 3) an upland landscape dominated by a sagebrush/bunchgrass/forb complex (see Cowardin et al., 1979).

The design goal for the wetland areas was to replicate, as closely as possible, the physical conditions found within the first two categories. The creation of this new wetland habitat was made possible by the need for suitable fill material outside of the riparian area. This fill was required in order to meet the elevation requirements for homesites within the valley flat and for the installation of utilities.

The first step in the construction process involved the use of scrapers to remove the two types of topsoil and place this material in different stockpile areas. Successive passes with the scrapers then removed over 30,000 cubic yards of gravel to create most of the wetland. Following excavation to design elevations, the topsoil was brought back to the site with the scrapers and spread out over the emergent and scrub-shrub wetland areas. The planting of native vegetation within the emergent margin and part of the scrub-shrub area was delayed until the construction of trout habitat within the stream

channel had been completed. All other habitat enhancement work, such as the construction of wood duck nest boxes, fox dens, etc., and revegetation efforts proceeded as excavation progressed. Typical design sections for this work are shown in Figures 1 and 2. The result was the creation of a wetland complex that would provide a wealth of wildlife habitat and serve as a barrier to the remainder of the Class A land within the property.

COSTS

The design and construction of the emergent and scrub-shrub wetlands, the relocation and reconstruction of Loggers Creek and the creation of the upper mile of trout stream was accomplished at a cost of about $470,000, which is equivalent to approximately $80/ft. This figure includes incidental costs, such as pre-project meetings with the city, environmental assessments, etc. Actual construction costs were lower, with channel construction averaging about $35 per foot and wetland construction averaging $20 per foot. The design and construction of the network of smaller trout streams within the River Run development cost about $210,000, which is equivalent to approximately $20/ft. By comparison, the average cost for the construction of channels designed solely for wild trout typically ranges between $10 and $15/ft less than the cost incurred here. The total cost for this work, which includes additional engineering work associated with the removal and reconstruction of the Loggers Creek headworks and other related projects yet to be completed, is anticipated to range between $675,000 and $700,000.

SUMMARY

The residential projects by the River Run Development Company in Boise, Idaho clearly involve an exceptional degree of concern for environmental quality. Nevertheless, these developments are indicative of the possibilities that arise when communities become actively involved with the planning process. Although past experience has shown that the habitat created here will not reach full potential for at least 6 more years (O'Brien, 1986), fish and wildlife will begin to utilize this area within the first year. Perhaps more significantly, the projects have demonstrated that fish and wildlife habitat does not have to be sacrificed as a result of changes in land use.

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Section A - A', Typical Tree and Shrub Species Planted to each Environment

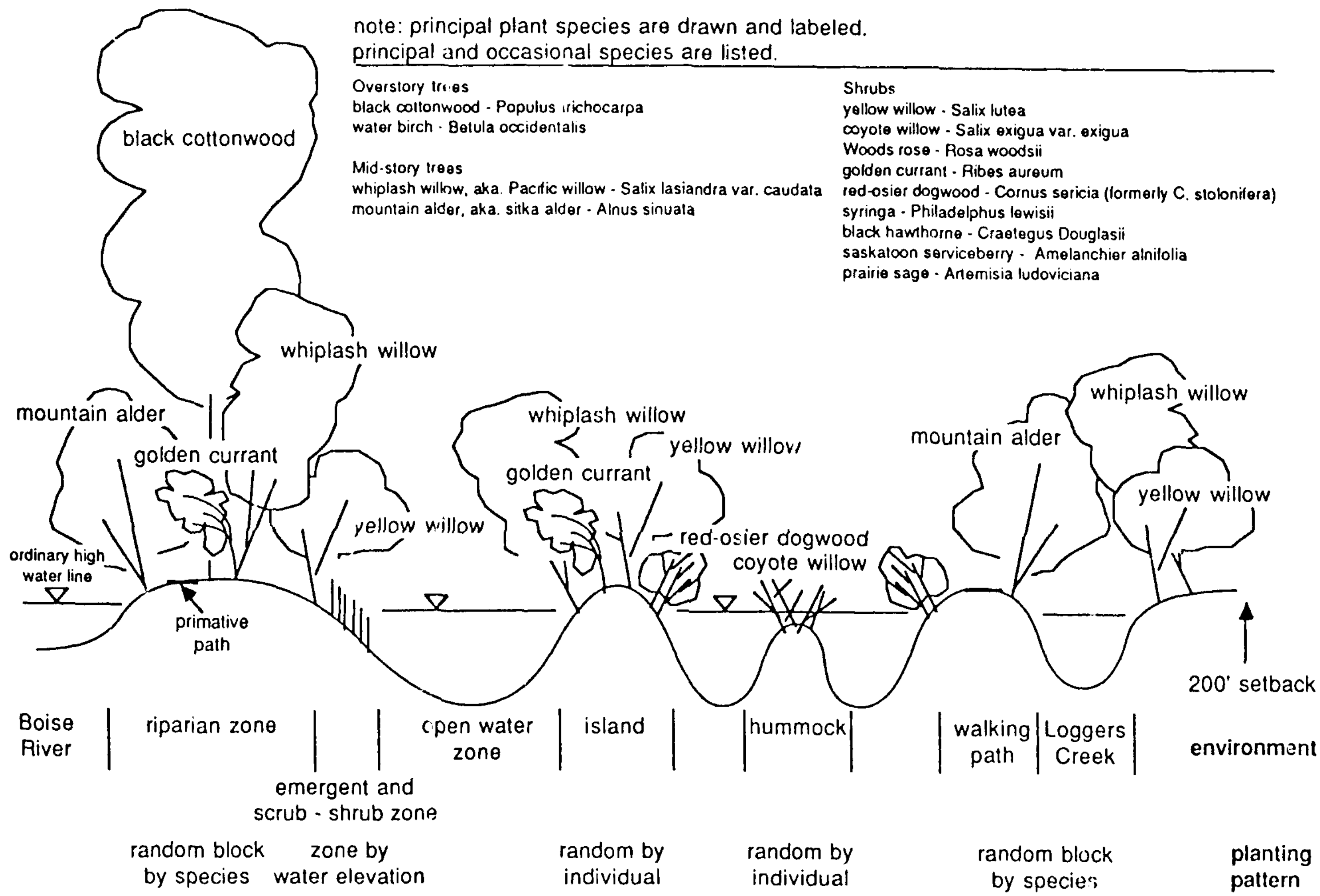


Figure 1

Section B - B', Typical Tree and Shrub Species Planted to each Environment

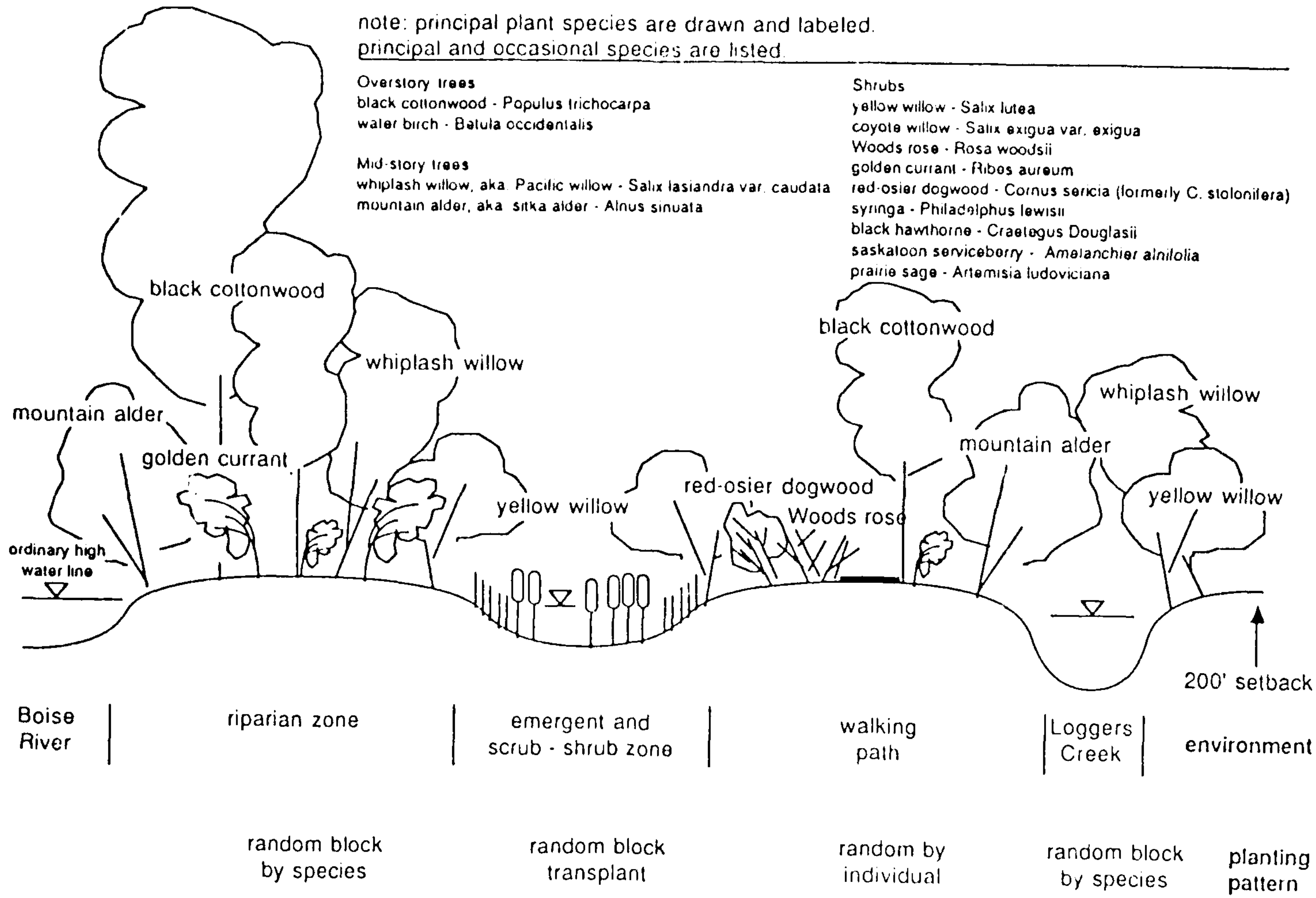


Figure 2

Hydraulic Design in Salt Marsh Restoration

Jeffrey Haltiner and Philip B. Williams
Philip Williams and Associates

INTRODUCTION

Since the latter part of the nineteenth century, over ninety percent of the saltmarsh along the California coast has been destroyed by diking and filling. Awareness of the ecological values of these ecosystems has prompted interest in methods to enhance degraded marshlands and create new ones. While biologists have been responsible for most of the research in wetland restoration and creation, a hydrologic and geomorphological understanding of the functioning of the physical components of the saltmarsh is critical to the productivity of the wetland. In particular, the hydrologic regime, in conjunction with the site topography and soils, is important. These features collectively create the physical environment to which the vegetation and wildlife must adapt.

Considering the narrow environmental bands in which specific wetland species thrive, it is evident that the wetland hydrologist must be capable of designing the hydraulic components of a marsh within relatively small tolerances. The design of the hydraulic components is dictated by the overall restoration goals, which include desired vegetation species, wildlife needs (cover, inundation tolerance, protection from intrusion), and fishery habitat. In addition, such factors as flood protection, navigation, access for construction and maintenance, and cost often must be included in the design process.

Our firm has been involved in many different types of tidal saltmarsh enhancement or restoration projects, including:

- -- Abandonment of a yacht harbor to allow reversion to marsh,
- -- Levee breach of a diked area to restore former wetlands,
- -- Introduction of limited tidal action to a flood basin during the dry season to enhance wetland values,
- -- Placement of dredge spoils to create suitable habitat,
- -- Excavation of dredge spoils overlying former marsh to restore wetland, and
- -- Excavation of upland to create tidal wetlands.

Restoration or enhancement of saltmarsh wetlands requires that many complex design criteria be interrelated. The most effective way of doing this is through the development of a marsh enhancement plan that determines the design, operation, and maintenance demands of the marsh. In developing this plan, a number of functions or potential functions of the hydrologic system have to be considered:

1. The hydraulic regime (the extent and period of inundation and circulation) needed to restore marsh vegetation and provide a suitable habitat for wildlife (provision of islands, "moats," etc.).

2. The sediment regime (long-term erosion or deposition throughout the marsh) which will determine the equilibrium configuration of the marsh.

3. The use of the wetland as a stormwater retention basin for flood control.

4. Other factors, such as the use of the wetland to improve the quality of stormwater runoff or sewage effluent.

The primary physical features of a tidal saltmarsh ecosystem include the estuary or coastal lagoon, intertidal mudflats, slough system, and the marshplain. Primary slough channels convey tidal waters between the marsh and estuary, while smaller secondary channels distribute the water within the marshplain. In a restoration project, these physical features may require modification to produce the hydraulic, sediment, and water quality patterns necessary to achieve the enhancement goals.

The remainder of this paper analyzes design criteria needs for the tidal slough channels, marsh plain, and hydraulic structures.

TIDAL SLOUGH CHANNEL DESIGN

In designing or modifying a tidal slough channel, the cross-sectional characteristics must be determined. In a natural slough channel, the geometry evolves over a long period of time in response to the net cumulative balance of erosion and sedimentation. Eventually, an "equilibrium geometry" is created that is dictated by the following parameters:

1. Tidal characteristics in the estuary, the most important of which is the tidal amplitude.

2. Potential tidal prism inland of the particular location in the slough channel. This is largely determined by the contributing marsh plain area and is analogous to a watershed in a river basin.

3. Sediment characteristics. The most important sediment characteristic is size, as it determines whether or not the sediments are cohesive. Sediment characteristics determine erosion and sedimentation rates.

4. Sediment concentration and siltation rates (as affected by sediment characteristics, salinity, and flow characteristics).

5. Biologic characteristics. Th[illegible] can have subtle effects on slough channel geometry. Different types of marsh plain vegetation influence tidal prism. Burrowing benthic organisms can increase erosion rate by loosening cohesive sediment.

Other factors are important in some instances. For example, the slough channel may also convey flood flows from an upland river. Such flow may deposit enormous amounts of sediment in the channels in a single event, creating non-equilibrium conditions.

In designing artificial slough channels, the balance between erosion and deposition has to be considered. If the channel is excavated too large for the particular upstream tidal prism, rapid siltation will follow. If it is constructed too small, erosion may take place if the sediments are sandy, but may be prevented where they are cohesive. In this case, the circulation and distribution of tidal waters to the marsh plain may be severely impaired. The relatively high cost of dredging precludes excessively deep or wide channels. However, some overdredging is usually prescribed to compensate for uncertainty in the final equilibrium depth.

Channel design for rivers or canals in alluvial sediments is typically undertaken using either "regime equations" or the method of "tractive force." Regime equations were initially developed by British engineers to aid in the design of irrigation canals in India. The method was later extended to natural rivers in the U.S. by Leopold and Maddock (1953), who referred to the analysis as hydraulic geometry. In this approach, channel characteristics such as width or depth are related to what is termed the "dominant discharge", that flow rate most important in shaping the channel cross-section. For river channels, this is generally accepted to be the discharge which occurs at the bankfill stage. An empirical approach is used in which cross-section geometry and dominant discharge are measured in equilibrium channels and the results plotted on log-log paper. The hydraulic geometry concept was applied to a tidal slough in the early 1960's (Myrick and Leopold, 1963), but little recent progress has been made.

During the past 10 years, our company has been involved in approximately 40 tidal marsh restoration projects. From this experience, we have developed hydraulic geometry relationships which provide a useful guide to the approximate channel dimensions for slough channel design.

Hydraulic geometry relationships are appropriate if a dominant independent variable can be identified and there exists uniformity in the channel-forming processes between different locations. In addition, to be useful, the independent variable must be easily measured. In alluvial rivers, the bankfull discharge has been identified as that flow which exerts sufficient bed shear stress to transport sediment, and occurs often enough to be of geomorphic significance. The bankfull discharge is either measured at a gaging station, or can be estimated using simple models.

For marsh channels, we have initially selected the "potential diurnal tidal prism" as the independent variable. This is the volume of water upstream of a channel cross-section which can potentially be exchanged between the marsh and estuary between the mean higher high (MHHW) and mean lower low (MLLW) tide elevations. This variable is related to the "bankfill discharge" concept, since the top of bank in most natural slough channels (i.e., the marshplain elevation) is close to MHHW for Pacific Coast saltmarshes. The potential tidal prism can be obtained by planimetering topographic maps (minimum 1-foot contour intervals) or by direct field surveying. In Figure 1, we have plotted potential tidal prism vs. marsh area for a number of California sites where adequate survey data exists. Since marsh area can be obtained very easily, this is a useful approximation for estimating tidal prism.

Figures 2, 3, and 4 present the relationships of maximum depth, top width, and cross-sectional area vs. potential tidal prism from four central and four southern California marshes which were considered to be in equilibrium condition.

Width/depth ratios are shown in Figure 5. Consistent with other hydraulic geometry relationships, the data are fit to an equation of the form:

$$x = ap^b$$

where x = dependent variable

p = potential diurnal tidal prism

a,b = regression coefficients

Such a relationship should plot as a straight

Figure 1

RELATIONSHIP BETWEEN MARSH AREA AND TIDAL PRISM DATA FROM VARIOUS LOCATIONS

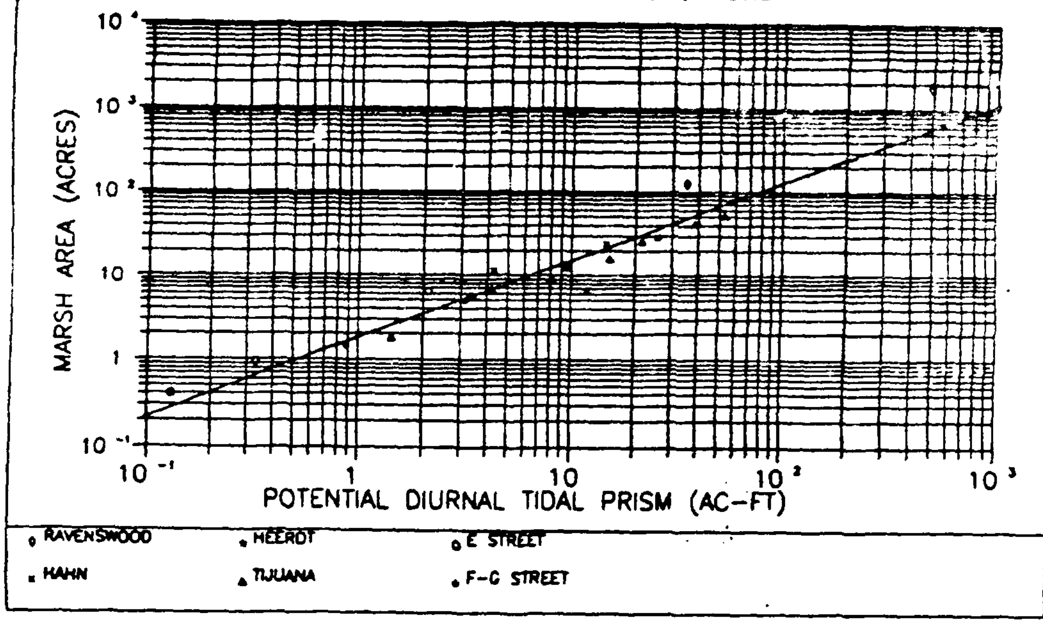
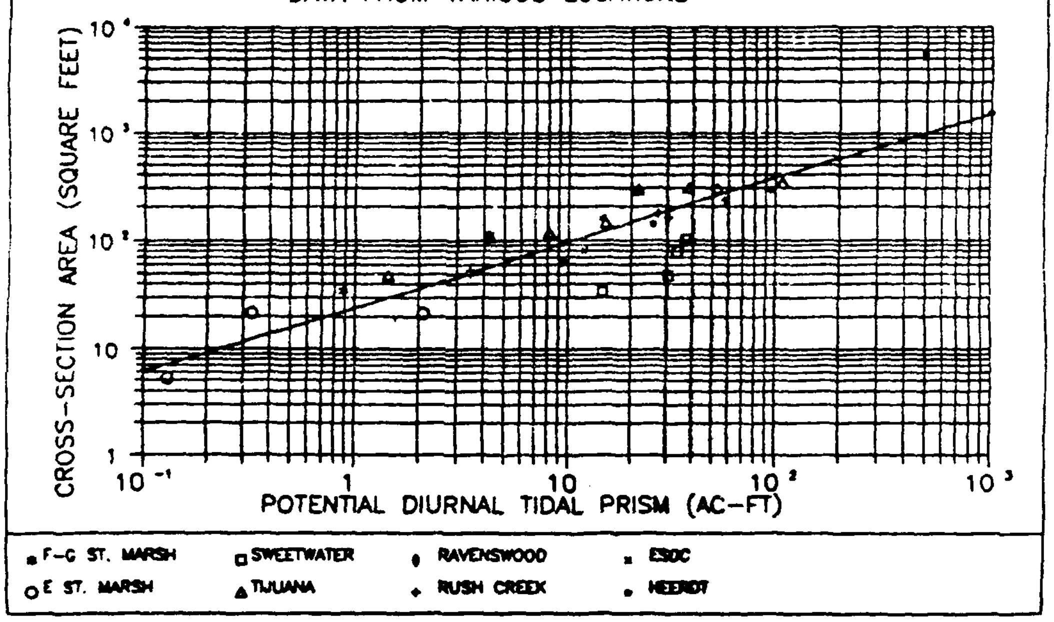


Figure 2

HYDRAULIC GEOMETRY RELATIONSHIPS: CROSS-SECTION AREA VS. TIDAL PRISM DATA FROM VARIOUS LOCATIONS



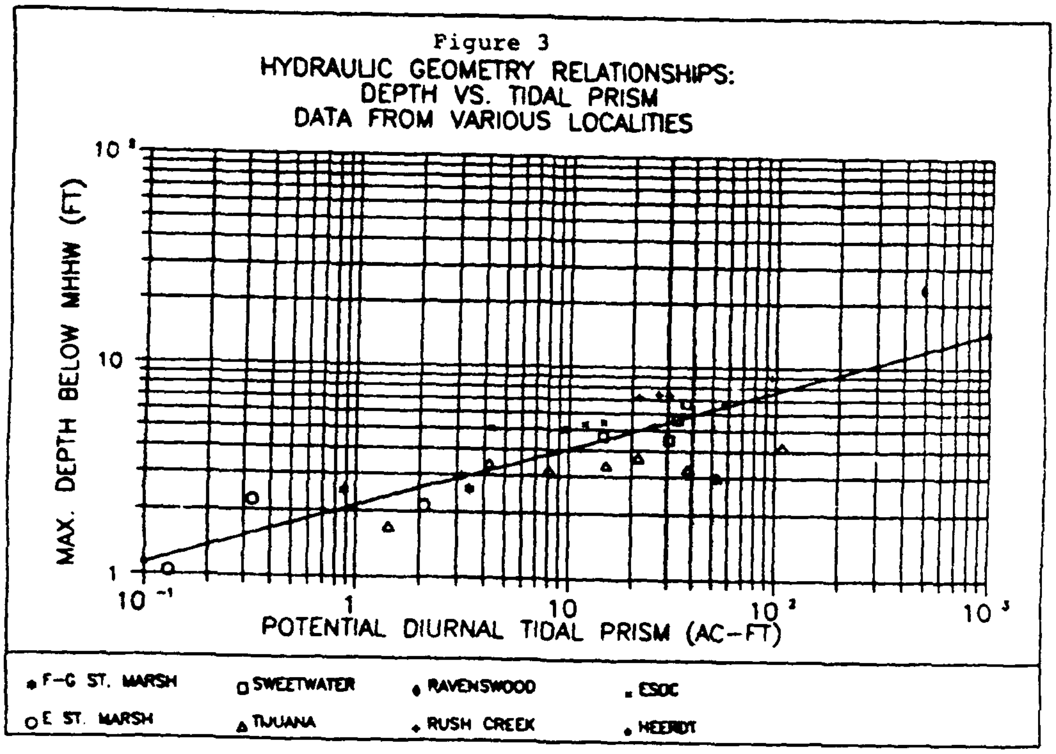


Figure 3
HYDRAULIC GEOMETRY RELATIONSHIPS:
DEPTH VS. TIDAL PRISM
DATA FROM VARIOUS LOCALITIES
MAX. DEPTH BELOW MHHW (FT)
POTENTIAL DIURNAL TIDAL PRISM (AC-FT)
F-G ST. MARSH
SWEETWATER
RAVENSWOOD
ESOC
E ST. MARSH
TIJUANA
RUSH CREEK
HEERDT

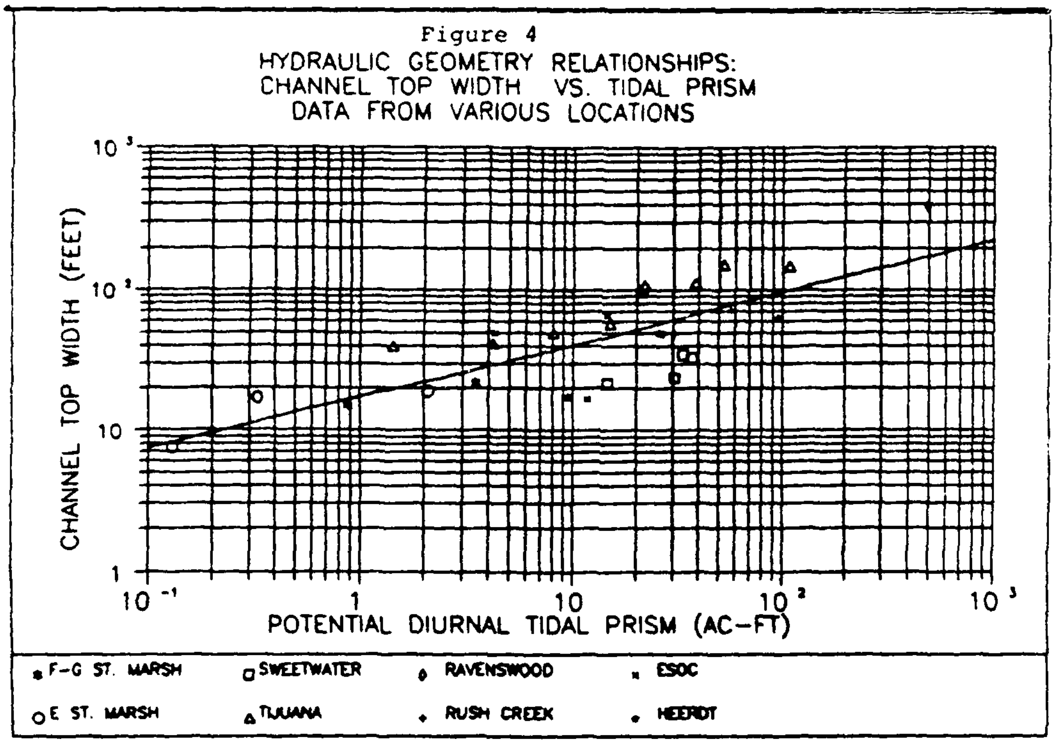


Figure 4
HYDRAULIC GEOMETRY RELATIONSHIPS:
CHANNEL TOP WIDTH VS. TIDAL PRISM
DATA FROM VARIOUS LOCATIONS
CHANNEL TOP WIDTH (FEET)
POTENTIAL DIURNAL TIDAL PRISM (AC-FT)
F-G ST. MARSH
SWEETWATER
RAVENSWOOD
ESOC
E ST. MARSH
TIJUANA
RUSH CREEK
HEERDT

line on log-log paper. A number of factors likely contribute to the scatter in the relationship shown. Channel sections were considered to be in "equilibrium" if neither the channel nor upstream marsh area had been significantly altered over a period of about 25 years. However, without long-term observations, it is impossible to tell if a channel is actually in equilibrium. It would of course be preferable to use data from undisturbed marsh areas, but relatively few of these remain and data is sparse. In general, it has been our observation that depth reacts much more rapidly to "changes" in tidal prism than does width. Secondly, while most of the salt marshes we have studied have occurred on predominantly fine-grained silt and clay sediments, a considerable range in the critical erosional and depositional stresses is likely, and this variation is not included in our relationships. A third uncertainty is the effect of "base level". In alluvial channels, a lowering of channel bottom (and consequently, the water surface) at the downstream end often initiates a head cut which migrates upstream, deepening and widening the channel until a new equilibrium occurs. This same process may affect slough channels if the adjoining estuary is dredged.

as the independent variable in alluvial river hydraulic geometry, is perhaps coincidental. The tidal slough channel will neither erode nor aggrade if the bottom shear stress is sufficient to resuspend the newly-deposited sediment but insufficient to erode the existing channel bottom. Data presented by Owen (1970) indicates that the density of the deposited sediment increases rapidly following deposition, and that periods as short as one day are sufficient to produce a significant increase compared to initial density. As the shear stress required to erode the sediment is directly proportional to density (Nicholson and O'Connor, 1986), it appears that the "channel forming" shear stresses should have a return period of no longer than a month, and possibly on the order of days or weeks. Since the peak shear stresses exerted by tides between MHHW and MLLW occur on the average once per day, the potential diurnal tidal prism appears to be an appropriate independent variable. However, the seasonal variation of tidal cycles in California produces about two weeks of higher-than-average cycles, followed by two weeks of lower-than-average cycles, and the effect of this is uncertain.

The choice of MHHW as the upper limit is

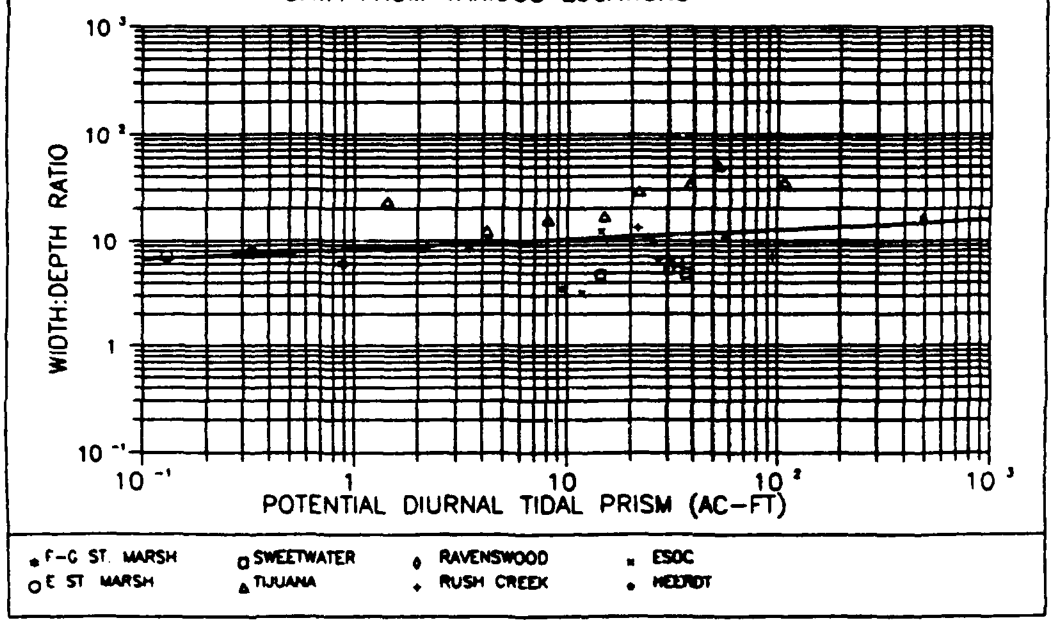


Figure 5
HYDRAULIC GEOMETRY RELATIONSHIPS:
WIDTH:DEPTH RATIO VS. TIDAL PRISM
DATA FROM VARIOUS LOCATIONS

The choice of the "potential diurnal tidal prism" as the independent variable was made based on frequency-magnitude considerations. That it corresponds to the bankfill stage in the slough channel, just as bankfull discharge is used

important with regard to marsh morphology. Since the marshplain is commonly at elevations close to MHHW, virtually all of the diurnal tidal prism is contained within the channels. At higher tides, the volume of the tidal prism increases rapidly as the entire marsh plain becomes

submerged. The effect of these greater flow amounts on channel formation/maintenance is unknown. It is also likely that measurement error accounts for some of the scatter in the relationships. While the dependent channel parameters (depth, width, and cross-sectional area) were surveyed, tidal prisms were estimated by interpolating between cross-sections.

MARSH PLAIN TOPOGRAPHY AND HYDRAULIC STRUCTURES

The most important design parameters to consider in marsh plain design are the tidal range and period of inundation. If the area has been diked, the level of this surface may have subsided several feet because of compaction or oxidation of the underlying clay/mud. Such an area, when subject to full tidal action, may be too low for marsh vegetation establishment. Low areas are more suitable for creating open water and mud flats. In other situations, the level of the marsh plain may have been artificially raised by filling or by the deposition of dredge spoils, resulting in excessively high elevations and soil salinity levels that prevent marsh plant survival.

There are two approaches to designing appropriate marsh elevations relative to the tidal range:

1. Modify the Topography. The desired tidal range can be established through topographic modification. The marsh plain should be graded to provide sufficient areas at the appropriate tidal elevations to promote establishment of desired marsh vegetation types. If subsidence has occurred, substantial amounts of suitable fill may have to be placed to raise the elevation. Thereafter, the marsh is subjected to full tidal action by breaching the surrounding levee.

2. Control the Inundation. If the marsh plain has experienced extensive subsidence, or if there are other constraints on the maximum tidal elevation, such as flooding in surrounding areas, tidal elevations can be controlled by using culverts, weirs, or pumps for tidal exchange with the estuary. This minimizes the grading required, and is desirable in areas where the original marsh drainage network still exists and is in the appropriate configuration for proper water exchange. This is often considerably less expensive to build, but requires continued maintenance and a reliable, knowledgeable management agency.

To determine what modifications will be needed to create conditions suitable for the desired biotic habitat, it is necessary to obtain the following information: (1) projected tidal range for the area; (2) accurate elevations for the area; (3) the location and amount of levee that will be removed or the size and location of culverts that will be installed; (4) the tidal prism and amount of damping (i.e., the amount by which the marsh tidal range is lessened relative to the estuary), if any, to the prism that will occur due to restriction of water entry; (5) the type and coverage of marsh plants desired; (6) wildlife factors; and (7) the amount and location of expected sedimentation and erosion.

If possible, tide characteristics should be measured at the site and cross-referenced to a nearby NOAA tide gage. If this is not feasible, nearby tidal data can be used and adjusted for any local conditions at the specific site as, for example, would occur if the marsh were at the end of a long, narrow channel. A typical design tidal cycle or sequence of cycles should then be developed. For example, a tidal cycle constructed from MHHW, MLLW, MLHW, and MLLW, according to the method outlined in NOAA tide tables, would provide a good indication of the range of tidal conditions to be expected.

The size of the levee breach or culvert opening affects both circulation and tidal range in the marsh. In general, it is preferable to have as wide an opening as possible. A calculation could be made of the difference between tidal range (water heights) in the estuary and the marsh. This calculation, called a "tidal routing computation," is made by estimating the flow through the opening for successive time increments. With this information, one can develop a tidal cycle inside the marsh for a typical tidal cycle in the Bay.

The flow through the opening for a given time increment is added or subtracted to the volume of water in the marsh. Using a volume/depth curve derived from a topographic survey of the marsh, the new water surface elevation in the marsh is estimated. We have developed a computer program which dynamically simulates water surface elevations in any number of marsh areas, interconnected by culverts, weirs, etc. as a function of estuarine tidal elevation.

Using this procedure, the damping of the tidal cycle for a given hydraulic structure or levee breach width can be estimated. The marsh vegetation species occupy a different range of elevations in a damped tidal cycle than in an unrestricted one. Either the marsh grading plan can be designed for a damped tidal cycle, or the levee breach can be made wide enough as to cause no significant reduction in tidal range.

The procedure described above can also estimate the tidal cycle at the end of a slough within the marsh. However, when the tidal drainage system is complex and channel frictional effects significant, the use of a computer model that simulates the flows (using the St. Venant equations) is required.

The first requirement in designing a grading plan for marsh plain restoration is an accurate topographic control on any grading carried out.

As has been shown in several San Francisco Bay marsh restorations, errors in grading of even 6 inches can greatly affect the type of vegetation ultimately established. Elevation requirements for marsh vegetation must be established locally.

In its natural state, the marsh plain is flat and drained by a complex network of tidal drainage channels. If the original marsh plain has remained undisturbed, subsidence has not occurred, or tidal inundation is to be mechanically controlled to the appropriate elevations, then filling is not necessary. If filling is required, however, there are three primary considerations:

-- There must be a precise elevation on the new, graded marsh plain surface;

-- A tidal network should also be graded; and

-- The fill material should preferably be soft estuarine mud. Other types of fill are generally unsuitable. Use of firm or stiff estuarine mud excavated from surrounding areas may cause difficulty for marsh plant restoration due to consolidation.

It may be desirable to create islands or preserve portions of dikes as waterfowl and wildlife habitats. Islands should be designed with elevation above the maximum predicted tide. However, more study is needed in these areas, as it appears that islands too high above the water surface are not used because of differences in vegetation. Slopes should be less than about 4:1. Soils should be suitable for growth of upland vegetation. The above configurations are more conveniently accomplished if grading is done before water is admitted. Once an area is wet, earth-moving equipment is less easily supported.

For those situations where the tide levels in the marsh are to be actively managed, an automatic slide gate controlled by a water level sensor may be suitable where electrical power is available. Where electrical power is not available, a gravity controlled flap gate can be designed. Trash barriers must always be included in the design to prevent debris from obstructing the gate. Furthermore, it is necessary to provide a back-up flap tide gate in case the control gate jams. The constricting effect of the control gate itself and of the approach culvert must be considered in analyzing the tidal hydraulics on the marsh site. The ongoing operation, maintenance, and monitoring requirements of an entire system should not be underestimated.

A degraded marsh may not retain the pre-existing mature internal tidal circulation system. In most instances, substantial modification and sedimentation have taken place. In order to create a new tidal drainage system, there are two approaches that can be taken.

One approach is simply to allow the tidal flows to establish a new drainage system by deposition and erosion. The problem with this approach is that a great amount of time is required for sloughs and channels to develop. Furthermore, where significant consolidation has taken place in the muds, erosion rates can be minimal, as has been observed in the Muzzi Marsh restoration. These factors can result in poor circulation and can delay the establishment of marsh vegetation.

A second approach is to create a new tidal drainage system of secondary channels. This must be designed to provide adequate circulation, velocity, and distribution of tidal flows within the marsh. These smaller channels, tributary to the major slough channels described earlier, have steeper gradients and have appreciable flow velocities only during the ebb tide. This network of smaller channels is of major importance in effective drainage of the marsh plain.

The drainage network should be designed so that no point on the marsh plain is farther than about 100 feet from a channel or slough. Drainage channels and sloughs should therefore meander in a similar pattern to the natural system to cover the largest drainage area in the shortest length. As observed in natural systems, slough junctions should be at roughly 120 degrees. Junctions of channels with sloughs should be roughly at right angles.

If possible, channels and sloughs should be laid out to create islands within the marsh plain. In addition to providing drainage and added diversity, such channels can act as barriers to prevent human and feral animal intrusion around the marsh perimeter.

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Utilizing a Tire Reef and Vegetation for Restoration of a Backwater Lake Habitat

Donald Roseboom, Richard Twait, and Thomas Hill
Illinois State Water Survey

INTRODUCTION

Much of the Illinois River flows through a wide, oversized, floodplain which results in a narrow channel flanked by numerous backwater lakes and sloughs. The lake, slough, and stream margin areas were once populated by large numbers of aquatic macrophytes. This type of habitat supported a rich and diverse fishery and wildlife resource.

By the mid 1950's, aquatic macrophytes had almost completely disappeared from the system. This was accompanied by a precipitous decline in the diversity and abundance of the wildlife supported by the plants (Mills et al. 1966, Bellrose et al. 1979).

Increased sedimentation of the river and backwater lake system occurred concurrently with the loss of macrophytes (Bellrose et al., 1983). Deeper areas in backwater lakes filled most rapidly (Demissie and Bhowmik, 1985), so that the bottoms of the lakes assume a shallow bowl shape. A limited amount of bottom structure is provided by submerged logs. The shape of the lake bottoms accentuate wind and barge induced disturbances which resuspend some of the bottom sediment (Jackson and Starret, 1959).

Material suspended in the water column greatly reduces the amount of light available to submersed and pre-emergent plants. Feeding success of many important species of gamefish is lowered because of their reliance upon sight to locate prey as well as the reduction or elimination of prey species within this environment. Material settling out of the water column can cover and kill fish eggs as well as bury the vegetative beds (Judy et al., 1984).

The fish populations which can withstand such conditions lack diversity, consisting mainly of carp, gizzard shad and channel catfish. Channel catfish are usually associated with submerged logs, the few remaining structural features of the lake bottoms.

The Peoria Lake restoration project is an attempt to develop low cost methods for restoring vegetative habitat in backwater areas. The project is being performed by the Water Quality Section of the Illinois State Water Survey for the Illinois Department of Conservation, Division of Fisheries. Wallop-Breaux funds underwrite the project.

STUDY SITE

The site chosen for the project is a bay located on the upper end of lower Peoria Lake near Illinois River Mile 166.0 (fig. 1). The bay is shallow, with an average depth of 1 1/2 to 2 feet during normal pool levels. The long fetch of Lower Peoria Lake adds to the problem of wind and wave disturbances resuspending some of the fluid bottom sediments.

A barrier island is located on the bay. A remnant stand of macrophytes exists behind this island. Broadleaf arrowhead (Sagittaria latifolia sp.) is one of the dominant species in the bed. This and another species of arrowhead, Sagittaria rigida (stiff arrowhead), were used in preliminary planting attempts.

Revegetation Attempts

Beginning in 1983, tubers of broadleaf and narrowleaf arrowhead were obtained from a commercial nursery. Naturally occurring tubers and transplants of broadleaf arrowhead from behind the island were also planted in the open water areas of the bay.

Initial plantings were, for the most part, unsuccessful. Some losses due to grazing occurred, but in most instances the plants were uprooted from the fluid sediments by wave action. The only plants which survived were protected from wind and wave action by a partially submerged log.

In order to protect the plantings from wave action, a 710 foot long breakwater was constructed from used tires. Details for breakwaters of this type are outlined in Schnick, et al. (1982). The breakwater was towed into the bay and secured to pilings to form an L-shaped configuration (fig. 2).

The north-south leg of the breakwater is approximately 460 feet long, with the remaining 250 feet running east toward the island. In addition to functioning as a wave energy dissipator, the assembly also functions as an artificial reef for fish habitat.

During late spring and early summer of 1987, 10,000 broadleaf arrowhead tubers were planted by hand in each of two areas: the unprotected open water and the area protected from wave action by the breakwater (fig. 3). Within six weeks after planting the arrowhead were beginning to emerge above the water surface.

enclosure placed in the unprotected area. Additional tuber plantings were made, along with transplants of broadleaf arrowhead from behind the barrier island.

By late August, all of the enclosed plantings and transplants had begun to flower. In the

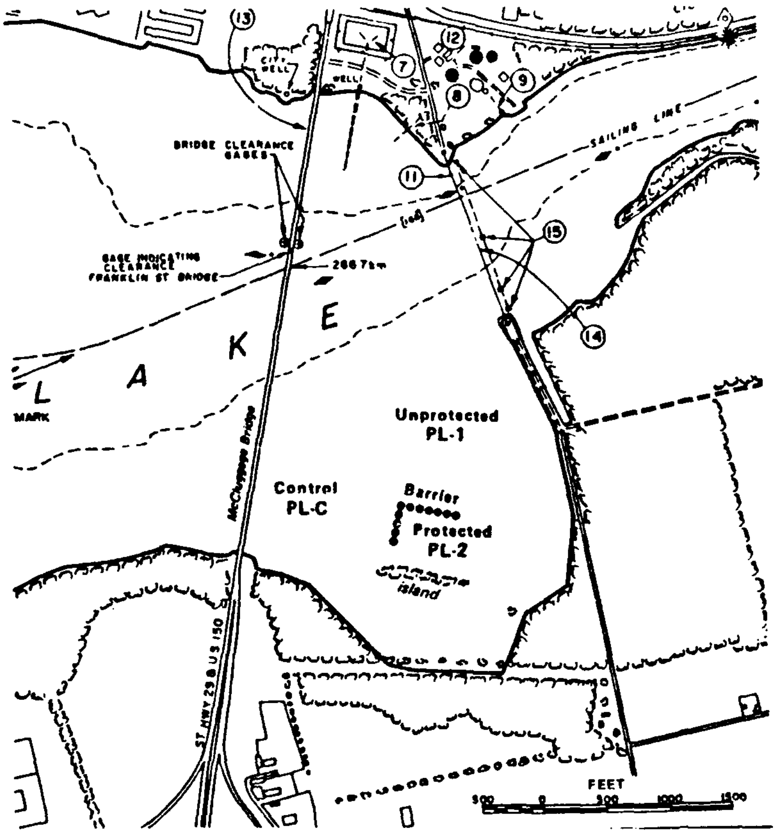

Figure 3. Schematic of study site showing planting areas and control area.

unprotected area, the only plants that survived to reach the flowering stage were in an enclosure placed near a partially buried log. The log appeared to provide partial protection from wave action. Tuber plantings and transplants in the other unprotected enclosure were uprooted within days after planting. Losses due to uprooting of plants were low for plantings behind the breakwater (fig. 4).

At the time that the plants were emerging, a flock of approximately 40 Canada geese inhabited the study site. One pair established a nest and all of the seven eggs successfully hatched.

In order to determine the effects of grazing and disturbance by rough fish upon the plantings, we had built a 25' by 25' enclosure with wooden stakes and poultry netting at each of the two planting sites. The grazing pressure exerted by the geese proved to be much greater than expected. In less than two days, all of the unenclosed plantings behind the breakwater had been cropped to the sediment surface.

Nine additional enclosures were then installed behind the breakwater with another

RESEARCH NEEDS

A major test for the effectiveness of the revegetation techniques used in this project will be the number of plants which reemerge in the spring. Successful overwintering of tubers will be

Figure 1. Aerial view of Lower Peoria Lake. Study site is located in the bay at lower right hand (northeast) corner of photo.

Figure 2. Breakwater assembly on shore immediately prior to emplacement.

affected by the amount of anchorage provided by the previous season's root growth in the soft sediments.

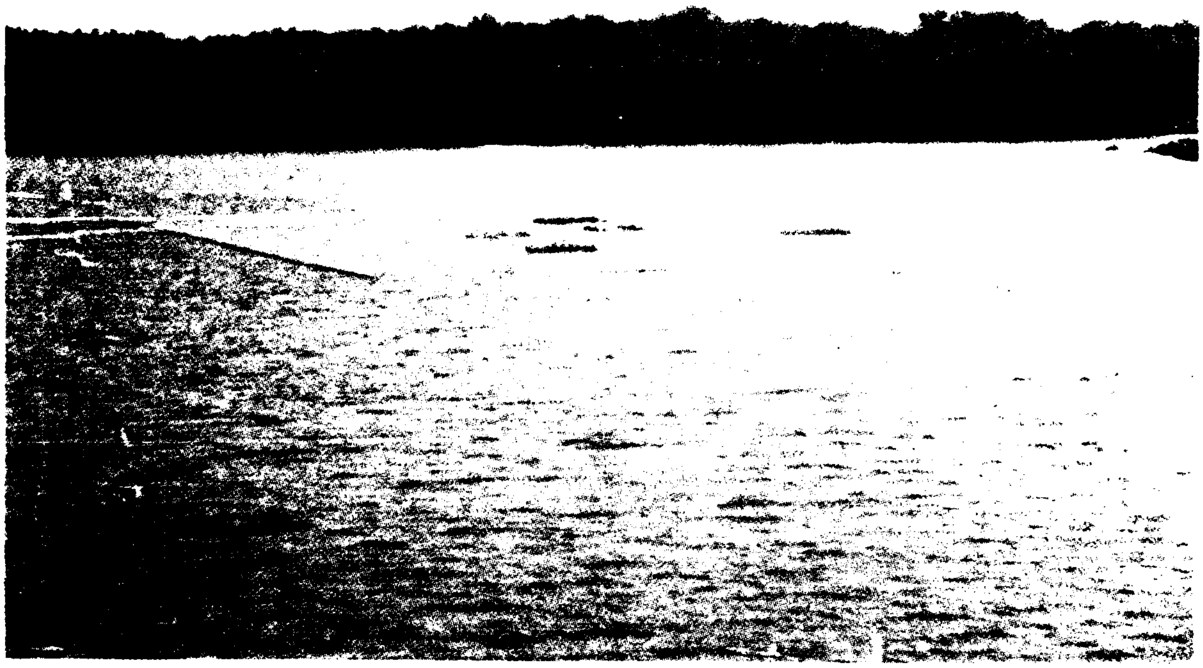

Figure 4. Photograph showing breakwater and protected plantings.

Weekly monitoring of water quality has been performed at the site since July. 1986. Monthly sampling of benthic macroinvertebrates and water column algal types and densities is also being performed. Fish population surveys are conducted by Illinois Department of Conservation streams biologists three times per year.

During a July, 1987 fish survey a largemouth bass and three other species not collected previously were taken from the vegetated site behind the breakwater. Continued increases in diversity and abundance of game fish and their young of the year, forage fish and macroinvertebrates will be the main criteria for success of this effort to reestablish aquatic vegetative habitat.

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chapter ten

Hydrology and Legal Issues

Hydrology, Wetlands Systems, and The Law

Alexandra D. Dawson, Esq.
Massachusetts Association of Conservation Commissions, Inc.

INTRODUCTION

Law and science rarely dine together. The problem is not so much distaste as difference in upbringing: They do not laugh at the same jokes, and they have very different ideas about the meaning of the word "fact". Nevertheless, the law of the land is supposed tc reflect truth and justice. Therefore, if there is sufficient scientific data on a particular point, e.g., that chlordane is poisonous, the law should eventually reflect that scientific consensus, e.g., ban the use of chlordane.

Statutes rarely reflect science directly. In fact, a statute explicitly banning use of a chemical, or increasing or decreasing the size of a wetland buffer, usually is viewed by the legal community as indicative of a breakdown in the system. Generally, statutes are supposed to establish frameworks for letting the science into the legal world. Thus, wetland protection laws broadly direct government agencies to preserve wetlands based on their importance in the hydrologic cycle. It is through administrative regulations that scientists mediate their information to legislatures and courts, and sometimes to the very agencies promulgating the regulations. This is how scientific information becomes fact in the legal world. Thus, 100 years ago, it was legal fact that groundwater was some mysterious kind of isolated underground river whose course no one could predict. Now, it is legal fact that ground and surface water are integrally and understandably connected.

In some areas of environmental protection such as chemical pollution, science is moving the law. In the area of wetlands protection, however, scientific fact has arrived late on the scene to confirm the intuition of early preservationists, whose efforts to preserve wetlands probably derived more from concern with wildlife and fish than from precise knowledge about the hydrologic system. Now that science is catching up with the minority perception of value, the general public (at least in the Northeast) concedes that wetlands are good things. This new value, in turn, is reflected in statutes and court decisions; and the actual implementation of the system of wetland protection will lean heavily upon administrative regulations. The use of such regulations is our way of putting new wine into old bottles: a drop at a time.

FEDERAL COURTS

In *United States v. Riverside Bayview Homes, Inc.*, 106 S. Ct. 455 (1985), the U.S. Supreme Court took its first look at 404 of the Clean Water Act, which directs the U.S. Army Corps of Engineers to establish a permit program to regulate certain activities in certain wetlands. In response, the Corps has promulgated extensive regulations. Like a landlubber exploring a sailboat in full career, the Court clung closely to the regulations and their history. In this case, the regulations were quite clear. Since 1977, the Corps has defined wetlands as "...those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions..."[1] Since the respondent's property admittedly matched this description, the only real issue was whether the Clean Water Act, designed to refurbish the quality of the waters of the U.S., could legitimately be extended by regulations to cover wetlands adjacent to, but not necessarily inundated by, surface water bodies.

Moving from rope to rope, the Court first remarked that "the question of whether Corps of Engineers may demand that respondent obtain a permit before placing fill material on its property is primarily one of the regulatory and statutory interpretation...." Second familiar handhold: "An agency's construction of a statute it is charged with enforcing is entitled to deference if it is reasonable and not in conflict with the expressed intent of Congress." An easy move to the bow: Reasonableness will be determined "in light of the language, policies and legislative history of the Act..." In reviewing the Act's congressional history, the Court determined that "the evident breadth of congressional concern for protection of water quality and aquatic ecosystems suggests that it is reasonable for the Corps to interpret the term "waters" to encompass wetlands adjacent to waters..." There follows a scant page about the science upon which the Corps quoting from the preamble to the 1977 regulations:

> "The regulation of activities that cause water pollution cannot rely on...artificial lines...but must focus on all waters that together form the entire aquatic system. Water moves in hydrologic cycles, and the pollution of this part of the aquatic system, regardless of whether it is above or below an ordinary high water mark, or mean high tide line, will affect the water quality of the other waters within that aquatic system."

Hanging onto the bowline, the Court was not about to say that all this is unreasonable.

Finally, the Court summarized the Corps' regulatory position: that wetlands may affect surface water quality even when they are not inundated by surface water. Wetlands may drain into surface water, and in so doing may filter and purify water, prevent erosion and flooding, and serve as valuable habitat. In short, the Corps had reasonably concluded that wetlands adjacent to surface water "may function as integral parts of the aquatic environment" and therefore may regulate their filling.

Seen in this light, the Court has now made it a legal "fact" that wetlands perform important hydrologic functions. This does not, however, mean that the Court believes that Congress must protect the nation's wetlands; only that it may, and has moved to do so in enacting the Clean Water Act. Nor it is likely that the Court would have recognized the hydrologic "facts" if the Corps had chosen to believe that wetlands not inundated by surface water somehow are less valuable for water quality maintenance (or, more likely, has determined that more research was needed before any conclusion could be drawn).[2] What this decision does is bring some legitimacy in popular and legal circles to the science reflected in the 404 regulations.

STATE COURTS

On the state level, the history of the Massachusetts Wetlands Protection Act (WPA) is illustrative of the same slow process of backing into science through the administrative process. The state's first coastal law, passed in 1963, apparently was based on a brief bylaw of a Cape Cod town requiring that "no one shall fill a wetland without a permit from the selectmen." From this simple beginning evolved a six-page statute regulating alteration of wetlands, together with 150 pages of regulations, 6,000 permit applications a year, and a host of legal decisions.

The WPA has never had a word to say about wetlands hydrology. In fact the law itself, long as it is, never states that wetlands are valuable or that they ought to be preserved. Instead, it provides that if a local agency, upon application, finds that a particular wetland is "significant" to any of eight statutory values, it is to "condition" the permit in such a way as to minimize damage to the stated value(s).

It was not until the late 1970s that science was applied to the legal structure. Prior to that time, the state wetlands regulations largely were procedural. That fact, combined with the statutory language, led to poor enforcement because the local permitting agencies: (1) lacked guidance on how to decide whether a wetland is or is not "significant"; and (2) believed that they could only condition, and not deny, a permit. In 1979, the state issued new regulations for coastal wetlands, which in 1983 were applied to inland wetlands as well.[3] (The early application to coastal wetlands, in part, was based upon scientific evidence of the special values of coastal marshes to shellfish and fisheries.) These regulations brought science into the regulatory scheme by main force. As published in 1983, the preamble sets forth the values of wetlands and the state policy of protecting them. Such a preamble, although not legally part of the regulations, may be cited in administrative or judicial decisions as indicative of regulatory intent.

More important, the regulations establish a series of science-based presumptions about the values associated with the different areas subject to the Act, which protects floodplains, beaches, flats, banks, dunes, and land under water, as well as various types of wetlands. They presume that some areas have only a few values.

However, certain wetlands (known as bordering vegetated wetlands or BVWs) are presumed valuable to all eight statutory values: water supply, groundwater supply, flood control, storm damage prevention, prevention of pollution, protection of fisheries and land containing shellfish, and protection of wildlife habitat.[4] The regulations go on to clarify that, if an applicant does not offer convincing evidence to rebut every applicable presumption, he or she cannot be permitted to make any major alteration to BVWs. Thus, the burden of proof is established in a manner understandable to lay permit agencies. Arguably, the applicant always had the burden of demonstrating that the proposed alteration would not adversely affect the wetland. However, since the burden of overcoming eight separate presumptions is so heavy, the regulations in many cases serve to prevent most alterations of BVWs.

There are, of course, holes in the system which demonstrate that scientific and legal "fact" are not yet identical. The major exemptions from the permit requirement are found in the statute itself, and are all politically motivated. For example, the WPA exempts alteration of wetlands as part of farming purposes. However, the stultifying presumptions are not applicable to an application for such conversion, which may be permitted by local or state agencies. This preference for farming is not scientifically logical, in the view of the great expansion of the cranberry industry in southeastern Massachusetts and the demonstrated negative effect of modern cranberry bog management on adjacent groundwater, wetlands, and estuaries (and probably human health).

Further, the statute gives only limited protection to isolated wetlands which are not adjacent to surface water. The presumptions treat these areas as important mainly to flood control, even though evidence exists as to other connections with the

hydrologic system. The distinction has been in the statute since its early formation: Wetlands are to be protected only if they "border on" surface water. The common perception is that this distinction is based on scientific evidence as to the lesser importance of isolated wetlands. It seems more likely, however, that the wording of the law is based on a conservative legal interpretation, also reflected in the Clean Water Act, that government may regulate surface water but not groundwater. Hence, wetlands, to be considered part of a regulatable system, must be associated with surface water. Until recently, Massachusetts had subscribed to the English common law rule of private ownership of groundwater, which may have influenced the legislative perception.

In the area of isolated wetlands, law has clearly yet to feel the imprint of science. Considering the proven importance of bogs and kettle holes to the New England hydrologic system (not to mention prairie potholes in the midwest), we can expect a slow and steady effort to build up the scientific substrate to change the law.

*This paper is reprinted from an article which appeared in the *National Wetlands Newsletter*, Mar-Apr., 1987.

NOTES

1. C.F.R. 323.2(c) (1978). The 404 regulations were revised in 1982 and again in 1986, but the definition of wetlands has remained unchanged.

2. It is also important to note that the Court did not set aside the taking issue. The decision clearly holds that a permit applicant may bring an action for compensation if the Corps permit denial "will prevent economically viable uses of the property or frustrate reasonable investment backed expectations."

3. Mass Admin. Code TIT. 310, 10.00 (1983).

4. Actually, the last value was added in 1986, and the regulations have not yet been amended to include it; however, there is no doubt that BVWs will be presumed valuable for this function.

The U.S. Supreme Court and the Taking Issue

Jon Kusler, Esq.
Association of State Wetland Managers

Edward Thomas, Esq.
Federal Emergency Management Agency

INTRODUCTION

This term the U.S. Supreme Court decided three cases in which land use regulations were alleged to be a taking of property. In all three cases, landowners argued that the regulations violated that portion of the Fifth Amendment to the U.S. Constitution which provides, "private property [shall] not be taken for public use without just compensation." Of the three cases the one most significant to land use planning--Keystone Bituminous Coal Association v. DeBenedicts (Keystone)--has been generally ignored by the press. The two other cases--Nollan v. California Coastal Commission (Nollan), and First Evangelical Lutheran Church of Glendale v. Los Angeles (Lutherglen)--were widely and erroneously reported as significantly curtailing the ability of state and local governments to regulate land uses.

What did the cases decide? What did they change? What actions should wetlands managers take in light of the decisions?

KEYSTONE COAL--FACTS AND HOLDING

In 1966, Pennsylvania adopted legislation which prohibited the mining of coal if that mining caused subsidence of residences, public buildings, or cemeteries. The articulated reason for this regulation was to protect the health, safety, and general welfare of the public. Several coal companies sued the state in 1982 alleging that the Pennsylvania law and regulations were an illegal "taking" of their property right to mine coal. The U.S. Supreme Court held that the value of the coal companies' property was not so substantially reduced as to become an unconstitutional taking despite the impact of the regulation in preventing the removal of some 27 millions tons of coal, constituting 50% of the available coal in some circumstances. The Court cited a long list of Supreme Court decisions over the last 70 years upholding highly restrictive regulations where issues of public health, safety or prevention of nuisance were involved.

This case validated state regulations almost identical to those overturned by the Court in the 1922 decision, Pennsylvania Coal v. Mahon, a case which has heretofore been viewed as one of the classic text book statements defining the outer limits of government power to regulate land. In Keystone, the Court emphasized the fact that while the state regulations might be similar to those overturned in Pennsylvania Coal, the purpose for which the new rules were adopted was to serve a community-wide need--unlike the previous rules, which were designed to benefit individuals. The holding in Keystone strongly endorses regulations which substantially reduce a landowner's property values where the regulations serve the important goals of protecting health and safety and nuisance prevention. Although this case dealt with regulations addressing a relatively uncommon hazard (subsidence), the rationale of the Court applies equally to wetlands water quality flood damage prevention and other types of hazard-reduction regulations.

THE NOLLAN CASE--FACTS AND HOLDING

The Nollans requested permission from the California Coastal Commission in 1978 to replace a small bungalow on their oceanfront property with a much larger house. The Coastal Commission approved the permit subject to the condition that the Nollans grant an easement to the public to pass laterally across their beach. The Commission's rationale for requiring the easement was essentially that the expansion of the Nollan's house would block the public view of the beach from the street, thus creating a psychological barrier to the public's use of the beach, and that the expansion of the Nollan's property along with other development in the area would increase private use of the beach, thus burdening the public ability to walk along the shore.

The Supreme Court indicated that the outright taking of an uncompensated easement would be the unconstitutional taking of property in this context. However, the Court stated that conditioning a building permit on the granting of such an easement would be permissible provided

the government regulation passed a two-part test: 1) the regulation must advance a substantial government interest, and 2) not deny the owner economically viable use of his land. In the Nollan case, the Court indicated that the requirement conditioning the Nollan building permit on their grant of an easement along the beach essentially advanced none of the purposes articulated for the requirement. The Court stated:

> It is quite impossible to understand how a requirement that people already on the public beaches being able to walk across the Nollan's property reduces any obstacles to viewing the beach created by the new house. It is also impossible to understand how it lowers the 'psychological barrier' to using the public beaches, or how it helps remedy any additional congestion on them. . . .

The Court specifically noted that it would have been constitutionally permissible to condition the permit on height restrictions, width restrictions, a ban on fences, or even the requirement that the Nollans provide a viewing spot on the property for passers by.

The case involved a unique set of facts in that outright public use was desired for the dedicated land, and there was little apparent relationship between the restriction and the stated goals of the regulations. In contrast, wetlands regulations which require setbacks, dedication of drainageways, "onsite detention areas," or "fees in lieu of" installation of storm drains or detention areas are clearly related to hazard reduction goals and do not permit public use of private land. However, the case does indicate an increased willingness of the Court to scrutinize the public purpose served by regulations, and the connection between the regulation and the purpose.

LUTHERGLEN--FACTS AND HOLDING

In 1978, a catastrophic flood in Los Angeles destroyed Lutherglen, an outdoor recreation camp for handicapped children, owned by the First Evangelical Lutheran Church. Shortly thereafter, Los Angeles County adopted interim floodplain regulations prohibiting reconstruction and new construction in the floodplain. Twelve of Lutherglen's 21 acres were affected by the regulations, and the twelve affected acres contained the area in which all the camp's buildings had been located. First Lutheran Church sued Los Angeles County on several theories, including a request for money damages for an unconstitutional taking of property. The single issue reviewed by the Supreme Court was whether the California Courts had acted properly in summarily dismissing, without trial, that portion of the church's complaint which requested money damages for the period during which a taking might have occurred. California courts had established a rule that when a person alleged a taking of property, their only remedy was to request that the ordinance in question be overturned.

Lutherglen never went to trial on the facts, and the California courts decided neither the validity of the interim moratorium on construction in the area, nor the validity of a permanent ordinance permitting greater land use later by Los Angeles County. The U.S. Supreme Court held that First Lutheran Church was entitled to their day in court to litigate the suit for dollar damages if they could indeed show that they had been "deprived of all use of their property."

This case was of great interest to legal scholars who have long been fighting in law review articles about the propriety of the California rule that a plaintiff in a taking case could not seek dollar damages. The value of the case to a state or local official who is attempting to properly and safely regulate land use is less clear. The case in no way discusses whether the ordinance in question was a taking. As noted by Chief Justice Rehnquist in the majority decision:

> We have accordingly no occasion to decide whether the ordinance at issue actually denied appellant all use of its property or whether the county might avoid the conclusion that a compensable taking had occurred by establishing that the denial of all use was insulated as part of the State's authority to enact a safety regulation.

Although it is important, the case was not a landmark decision "for which the developers have been waiting." It is rather a highly technical case addressing a single, narrow issue--available remedies for taking. It is also a highly complicated case which raised more questions than it answered. The confusing nature of the decision and many unanswered questions detract substantially from its precedent value. However, because of widespread misunderstanding about the case, it seems destined to produce a considerable amount of confusion and litigation.

LESSONS OF THE CASES

Wetland managers can draw several lessons from these three cases, when viewed in the context of other Supreme Court and lower court decisions over a period of years.

- Regulations adopted for valid public purposes and with an adequate basis in fact may substantially reduce land values without effecting a "taking." Hazard-reduction regulations have universally been upheld as serving valid public purposes.
- The impact of regulations must be

evaluated for an entire piece of property (not just one portion) to determine whether a taking has occurred. This means that hazard-related setbacks which affect only portions of a property are quite clearly not a taking.

- Public safety and prevention of nuisance is a paramount concern of government and no landowner has a property right to threaten public safety or cause nuisances. Control or abatement of even existing uses has often been sustained to achieve these objectives.
- Regulations are a taking only if they deny all use or all economic use of an entire property, including reasonable "investment-backed expectations." Even then, regulations may be valid under certain circumstances where the only economic uses are nuisance-like.
- A moratorium on development may well be legally sustainable in most states if it is properly for a specific limited public purpose, sets a specific date for termination, and is caused by sudden need outside the control of government. However, in view of the fact that a moratorium was the subject of the Lutherglen Case, adoption of moratoria in the future will likely be a "red flag" provoking needless controversy and litigation. (See below for hints on avoiding moratoria problems).

In summary, there is little chance that hazard-related wetlands regulations will be held a taking despite this trilogy of cases from the Supreme Court. Performance-oriented regulations such as building codes, floodway restrictions, and grading codes are particularly unaffected. Nevertheless, the decisions indicate an increased willingness to examine the nexus between regulations and regulatory goals. Attempts to require actual public use of private land will continue to be viewed particularly closely. And these cases do increase the possibility that regulatory agencies will be sued by developers (even if they do not win), particularly if the regulations in question are unusual or highly restrictive.

In considering whether or not to regulate, governments must assess not only the potential of developers' suits, but also their own potential liability for increased hazard losses. If a government is faced with a decision about whether to adopt performance-oriented hazard regulations or not to regulate (with resulting damage to subdivisions, houses, and public works, and potential law suits at that time), the choice from a purely legal perspective is clear: regulate. There is every indication that hazard-reduction regulations will continue to be upheld. And local and state governments are held liable at an alarming rate for hazard losses due to government actions which increase hazards, or to inactions (failure to enforce regulations, to maintain storm sewers or dams, or to carry out necessary inspections).

ACTIONS TO TAKE IN LIGHT OF THE DECISIONS

First--Stay calm. View the cases for what they are. Remain confident that soundly conceived and fairly administered land use regulations will continue to be sustained by the courts. There have been hundreds of state and federal court decisions upholding hazard-related land use regulations. There have been very few to overturn such regulations. It is likely that even the prohibition on rebuilding which caused the controversy in Lutherglen will be sustained once the case goes to trial.

Second--Ask landowners or lawyers citing these cases if they have actually read the opinions. If they claim that these cases generally hold wetland, floodplain, or other land use regulations are unconstitutional, it is unlikely that they have read the opinions.

Third--If you, as a regulatory agency, wish to minimize the chance of "taking," emphasize performance standards in your land use regulations.

Fourth--Take the following normal precautions, which many of you have been using for years, to avoid a "taking," especially when land values are high and impacts on the landowner are particularly severe:

1) Provide a variance or "special permit" procedure in regulations, since such provisions are very rarely held to be a "taking" on their face, and they provide the regulatory agency with the opportunity to deal with extreme hardships.
2) Emphasize health and safety considerations and the prevention of nuisances, in your regulations and in your written findings for individual permit denials. Regulatory actions closely tied to these objectives are rarely held a "taking."
3) Link your regulations to national and state-wide programs such as the National Flood Insurance Program. Courts have been particularly willing to sustain such regulations.
4) Apply large lot zoning (two-ten acres) to area-wide land use restriction where appropriate or possible, since courts have held that regulations which permit some reasonable use on an entire property do not constitute a "taking."

5) Document with particular care the need for the regulations and the reasons for your permit denials in urban or other settings where land values are very high.

6) Encourage pre-application meetings by permittees so that mutually acceptable project designs can be formulated.

7) Apply your regulations in a consistent and equitable manner. Maximize the opportunity for notices and public hearings.

8) If you adopt a moratorium, do so for a fixed period and make sure that a) the reasons for it are clear and legitimate, and b) there is a viable variance procedure. Generally regulations which permit a developer to provide information needed by government decision-makers is preferable to a moratorium.

9) Coordinate regulatory, tax, and public works policies to insure that the fiscal burden on landowners for community services is consistent with permitted uses.

10) Apply, in extreme circumstances, transferable development rights to help relieve the burden on landowners.

11) Use acquisitions rather than regulation where active public use is needed for land, of where a single landowner or group of landowners must bear disproportionate burdens for the public good.

Fifth--remember that landowners have rights protected by the Constitution. Be fair, be reasonable, but protect the public interest above all. From a legal perspective not much has really changed. Be confident!

*This paper is an expansion and modification of an article which appeared in the "National Hazards Observer" in September 1987. The views expressed are those of the authors, and do not necessarily represent the view of any organization or agency.

chapter eleven

Multiobjective Management of Wetlands

Floodplain and Wetland Coordination

Edward A. Thomas, Esq.
Federal Emergency Management Agency

INTRODUCTION

Coordination between wetland protection programs and the National Flood Insurance Program offers a rare and wonderful opportunity for synergism in the public interest. Traditionally, most wetland protection programs have focused on water as a source of life for fish, wildlife, and humans. The National Flood Insurance Program, on the other hand, has a specific and narrow focus relating to water as a source of death and destruction. Given their differing reasons for existence, the two programs, at times, will be concerned with different issues and geographic areas. However, because both have their roots in a single substance--water--knowledge of and sensitivity to one program is likely to substantially enhance the proper management of the other program. Wetland managers are finding that the courts are generally more receptive to strict regulation based on public safety concerns such as flooding, rather than on more nebulous reasons of environmental protection. Floodplain managers increasingly find that destruction of natural wetland areas so interferes with the natural hydrology and hydraulics of some bodies of water as to create unanticipated and unacceptable risks. This article offers a brief introduction to the National Flood Insurance Program from the point of view of a wetland manager. It also highlights the Commonwealth of Massachusetts' successful coordination of wetland and floodplain management.

THE NATIONAL FLOOD INSURANCE PROGRAM

The federal government tried for years to keep floods away from people through the construction of dams, dikes, levees, and other flood prevention structures. After spending billions of dollars over more than a century, it became clear that, despite this massive public works effort, the nation's flood losses were increasing. Therefore, Congress in 1968 established the National Flood Insurance Program (NFIP). The primary purpose of the Program is to encourage state and local governments to adopt reasonable flood reduction measures governing development in "special flood hazard areas," which are those areas inundated by a flood that has a one percent annual chance occurrence, i.e., the 100-year flood. Such measures are to be in conformance with minimal federal standards promulgated by the Federal Insurance Administration (FIA). In addition, the Program provides an organized, cost-effective system of indemnifying victims of flooding. In 1973, Congress substantially strengthened the NFIP by mandating that all flood hazard areas be identified and that substantial sanctions be imposed on communities that choose not to adopt the local zoning, building code, or other measures to properly regulate construction in such areas. Since the vast majority of communities with developed land identified as flood hazard areas participate in the NFIP, most development in flood hazard areas is subject to the Program's minimum federal standards.

Floodplain Management

The NFIP is administrated in two phases: The Emergency Program, an interim phase scheduled to expire in 1990, and the Regular Program. To participate in the Emergency Program, the local government adopts standards regulating construction and substantial improvement in the flood hazard area, based on information provided by FIA's Flood Hazard Boundary Map or other local data. For communities with substantial flood hazards, FIA then conducts a detailed Flood Insurance Study of flood hazards which leads to the issuance of a Flood Insurance rate map more precisely delineating the flood boundaries and providing elevations of the "base" or 100-year flood along the major bodies of water.[1] There are detailed procedures for consultation with local officials and for appeals in order to provide proper due process in the conduct of the flood study.

Under the Regular Program, upon completion of the Flood Insurance Study, a municipality has six months to adopt detailed zoning ordinances, building codes, or similar measures to regulate within the flood hazard area the placement of fill, building construction, and substantial improvement of existing buildings. The minimum federal standards vary according to the type of flooding and of technical data. Generally, these standards require that new and substantially improved residential construction be elevated to or above the levels of the 100-year flood. Nonresidential construction must either be elevated or floodproofed so that the building is watertight to the level of the base of flood.[2]

The Floodway

Under the Regular Program, the community is required to designate a regulatory "floodway" along major rivers and streams. The floodway is that portion of the stream or river plus floodplain necessary to convey the waters of the 100-year flood without increasing the level of that flood more than one foot at any point.[3] Conceptually, it can be visualized as an undeveloped area serving as a sort of natural highway through which flood waters can move without substantially backing up. The federal standards effectively prohibit the placement of fill, buildings, or other obstructions within the floodway.

As part of the Flood Insurance Study, FIA will calculate a floodway based on an equal loss of conveyance from each side of the stream or river, unless the community requests some other method of calculation. The exact placement of floodway boundaries is, in reality, a political decision made within certain engineering parameters. Typically, a great variation in the exact placement of the floodway boundary is technically feasible. Wetland managers may wish to participate in the decision-making process to insure optimum protection of critical areas. Even in the many communities where floodways have been designated, the local governing body may request that FIA approve a change in the placement of the floodway boundary to meet community needs. In most cases, such requests are made in response to development pressures.

Enforcement

The local government is responsible for administering and enforcing its floodplain management, or loss reduction, standards. It may, in accordance with federal procedures, grant variances from those standards. However, improper issuance of variances or failure to properly enforce the Program's standards may result in the imposition of such federal sanctions as probation or suspension from the NFIP. In addition, FIA may pursue its common law right to subrogation when someone causes or exacerbates flood conditions, resulting in the payment of increased federal flood claims.

Purchase of Flood Damaged Property

The NFIP may also provide an opportunity for wetland managers to purchase developed wetlands. Buildings insured by the Program that are substantially destroyed or frequently damaged may, in some circumstances, be voluntarily sold to FIA, and turned over to a state or local agency for demolition and reuse as open space. At the present time, approximately $4.7 million is authorized annually for this aspect of the Program.

Program Limitations

The NFIP is a complex matrix of compromises between a person's constitutional right to own and make use of private property and the government's duty to protect public health and safety by avoiding cataclysmic destruction of property. It is important to recognize that the Program applies only to areas inundated by the 100-year flood and does not regulate development outside the flood hazard area. Even with the 100-year floodplain, the floodway standards permit at least some additional flood depth and consequent increase in the size of the area flooded. In addition, the natural flood storage capabilities of floodplains and wetlands are not included in calculating a floodway. Therefore, what might have been a shallow, slow-rising flood lasting several days may become a fast, deep flood which lasts a matter of hours.

It is difficult to generalize about the effects of losing the natural flood storage provided by wetlands. Some river systems experience catastrophic changes in flood depths and velocities when significant wetlands have been filled. Determining the precise effect of wetland fill activity on flooding requires complex hydraulic calculations concerning the size of the watershed, the total volume of unfilled floodplain land, and the percentage of wetland storage in the system, as well as a hydrologic determination of the total quantity of water which will be flowing through the watershed in a given flood. The filling of any wetland area or the precedent made by the filling of a particular wetland might be critical to the prevention of catastrophic flood losses.

FIA flood hazard maps also are limited in scope because they are designed to identify areas prone to disastrous flooding. Therefore, areas of comparatively minor flooding of one foot or less and flood areas with a drainage area of less than one square mile are not shown as flood areas. Given these limitations, state regulations more stringent than the minimum federal standards easily can be justified on several grounds of public safety. Such public safety-based regulations may have the happy effect of providing a legally sustainable basis for increased levels of wetland protection that could not be justified under a more traditional wetland protection rationale.

Executive Order 11988

This federal executive order severely constrains the issuance of federal grants and loans in flood hazard areas. It prohibits federal agencies from undertaking activities in floodplains unless they issue a public notice; determine that there is no practicable alternative to the proposed construction; and mitigate the harm caused by the activity. Federal agency failure to follow these procedures has resulted in the modification,

delay, or cancellation of a variety of housing developments and other buildings proposed for location in floodplains.

STATE INTEGRATION OF FLOODPLAIN AND WETLAND PROTECTION

The Commonwealth of Massachusetts has used the NFIP to assist in achieving both flood damage reduction and wetland protection. It has done so by closely integrating the Program's standards and maps into the state Coastal Zone Management process, Building Code, and Wetlands Protection Act. In addition, the Commonwealth has done a model job of using funds provided by the NFIP to acquire flood damaged properties on developed barrier beaches.

The Commonwealth's effort to coordinate its Wetlands Program with the NFIP is particularly noteworthy. Under the regulations implementing the Act [see *Hydrology, Wetlands Systems, and The Law*, Alexandra Dawson], areas subject to the 100-year flood as calculated by the Federal Emergency Management Agency are presumed to be wetlands. Performance standards governing work within the 100-year floodplain are more stringent than the NFIP standards and generally require compensatory storage, construction of flood retention basins to control increased runoff, and evacuation plans. The compensatory storage requirement serves to avoid most of the potential problems inherent in the loss of natural wetland flood storage. The Act also provides state-level oversight and technical assistance to the local officials who make the permitting decisions.

RECOMMENDATIONS FOR WETLAND/FLOODPLAIN COORDINATION

Cooperation and coordination between and floodplain managers at all levels of government can be mutually beneficial. Recommendations that may assist the wetlands manager in modifying a program to take advantage of the NFIP are:

(1) Recognize that the courts will be extremely supportive of regulations designed to protect public safety. In appropriate circumstances, explicitly indicate the public safety problems associated with a given development proposal involving wetlands destruction or degradation.

(2) Understand that the information on floods provided by the NFIP is valuable, but only one piece of the public's legitimate interest in wetland protection. The NFIP is a minimum. More rigorous wetland and floodplain protection easily can be justified for both environmental and public safety reasons.

(3) Recognize in your laws and regulations that any area with the 100-year floodplain is a wetland and, therefore, should be regulated as such.

(4) Perform, or require the developer to perform, the calculations needed to determine whether the filling of a particular wetland together with other similar anticipated fill activity is likely to result in significant increases in flooding potential.

(5) Develop a rapport with the state and federal floodplain managers who should be working with you to achieve related goals.

NOTES

1. Flood Insurance Rate Maps also delineate areas designated as undeveloped coastal barriers under the Coastal Barrier Resources Act. New construction or substantial improvement of existing structures in such areas are ineligible for most forms of federal assistance, including flood insurance.

2. See 44 C.F.R. 60.3(c), (d), and (e). The minimum federal standards require special regulations for the "coastal high hazard area," which is the strip of land along the coast subject to flooding and wave action. In this area, the federal standards prohibit the use of fill for structural support of buildings, as well as any manmade alterations of sand dunes and mangrove stands that would increase the flood hazard. Also, they require special construction measures.

3. Some states have adopted more restrictive standards that do not permit even a one-foot increase in flood depth, and consequent increase in the size of the flood hazard area.

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Planning for Multi-Purpose Use of Greenway Corridors

Ronald D. Flanagan
R.D. Flanagan and Associates

INTRODUCTION

A wetland scientist, flood management planner and Sierra Club member were hiking in the mountains. The three noticed, to their horror, a large bear charging down a ridgeline toward them. Seized by panic, the planner and Sierra Clubber noticed the wetland scientist sitting down and removing a pair of running shoes from his backpack. "What are you doing?" cried the panic-stricken planner, "You can't outrun that bear, it can run down a horse."

The wetland scientist calmly replied, "I don't have to outrun the bear, I just have to outrun you."

This story, sadly, illustrates the attitude environmentalists have had toward each other: each going his own separate way, taking care of his own individual concerns. When funds were more plentiful this attitude was possible, but it does not work any longer, especially when there is more than one bear coming down the hill. To survive, we must all work together.

We have seldom succeeded in attaining and maintaining a strong constituency for either floodplain management, wetlands protection, wildlife, or nature preservation, at the national or local level. The average citizen just doesn't give a damn.

U.S. floodplains are being developed, streams channelized, wetlands dried, habitat destroyed and natural areas urbanized because there is an economic incentive to do so. We just haven't found a way to make a buck protecting our environment--and in a capitalist society, that's the name of the game. Nor have we been very successful in involving the vast majority of the public in our concerns.

In a tight economy with limited budgets, unfortunately, environmental concerns are the first to be axed from the budget at all levels of government. There are just too few dollars to go around.

Times of crisis, however, often create opportunities and force us to think more cleverly. The most valuable wildlife habitat is found adjacent to streams. Most mature vegetative stands and scenic areas are located along waterways. Most wetlands are in floodplains. Vegetation cleanses the water and air of pollutants. Surveys have found that 80% of recreationists prefer to recreate near water features. Thus, our individual interests and concerns end up occurring in the same area: the floodplain. If we preserve and protect the floodplain, we also protect wild and scenic rivers, wetlands, wildlife habitat, fish, vegetation, and scenic vistas. We help reduce air and water pollution and urban flood damage, which now exceeds over $4 billion annually. We provide the public with linear parks, open spaces, and trail systems for nature observation, riding, hiking, walking, running and biking.

All of these worthwhile public objectives could be accomplished with each dollar expended, if we spent it wisely.

There are signs, however, that we are beginning to work together, and one of the best examples is the multi-disciplinary planning and cooperation taking place across the nation for multi-purpose use of critical open-space and greenway corridors along our nation's streams and rivers.

I want to summarize community programs which illustrate this concept:

-- Tulsa Trails (Oklahoma) has expanded the constituency base for flood plain management to include hikers and bikers with great success in achieving multi-purpose goals..

-- Rapid City, South Dakota, turned a disaster into a community asset and created a major greenway.

-- Garden City, Oklahoma has also shown how a disaster can afford an opportunity to accomplish many public objectives with each tax dollar.

This paper issues a call to arms, describes the need to form an environmental coalition for multi-purpose planning and protection of greenways, and finally calls for the creation of a computer-aided base mapping system for use nationwide.

TULSA TRAILS: A LESSON IN SYNERGY

In my home town, Tulsa, Oklahoma, we've

been beating our heads against the wall promoting flood plain management for more than 20 years. Despite the fact that Tulsa is among the nation's most flooded cities (the 100-year flood occurs, on the average, every four years), progress there has been painfully slow (Rubin, 1987).

Tulsa's Department of Stormwater Management is constructing maintenance trails along the city's creeks and streams. With very little additional cost, these maintenance trails can double as recreation trails for hikers and bikers. These corridors also serve as linear parks preserving the floodplain, providing wildlife habitat, and preserving mature vegetative communities. These corridors link neighborhoods, parks, schools, and community centers, providing safe places for families to bike and hike. (See figure 1.)

This concept has gained widespread community support from hiking, running and biking clubs, with thousands of members. Amazingly, the Tulsa Trails project is bringing us some important spinoff--synergistic--benefits. Citizens who never cared about floodplain management are now supporters because of their interests in trails. We were able to link our concerns with their interests.

If this can happen in Tulsa, it can happen anywhere in the nation.

RAPID CITY, SOUTH DAKOTA: DISASTER TO URBAN GREENWAY

In 1972 Rapid City, South Dakota, suffered a devastating flash flood on Rapid Creek that damaged $160 million in property, destroyed 1,200 buildings and killed 238 people. Through the leadership of Mayor Don Barnett, the community instituted a national prototype floodplain acquisition program, removed the damaged homes from the floodplain, and created a six-mile long, quarter-mile wide urban greenway open space through the center of the city. The greenway now contains parks, recreation trails and golf courses.

South Dakota is an outdoors, sports, and tourist-oriented state. In the design and reconstruction of Rapid Creek, the engineers were ingenious enough to create fish habitat along the banks. Rapid Creek was stocked with sports fish and is now the most popular sport stream in the state. Rapid City stands as a creative example of enlightened leadership turning a disaster into a multi-use community asset that benefits all aspects of the city, including commerce and the tourist industry.

GARDEN CITY, OKLAHOMA: MULTI-PURPOSE STORMWATER DETENTION POND

The Tulsa area suffered severe flooding from the Arkansas River in October 1986 which devastated the small area of Garden City on the Arkansas River floodplain. The neighborhood contains older, very moderately priced homes sandwiched between industry and oil refineries and had been designated for years by Urban Renewal for clearance and industrial redevelopment. Until the flooding and drainage problems were solved, however, there was no economical way to redevelop the area.

The Arkansas River flood changed all of that. The homes flooded an average of eight feet and most suffered such substantial damage they could not be rebuilt without elevating the structure above the flood heights. This was not economically feasible because none of the properties had flood insurance.

Using the flood disaster as an opportunity, the Department of Stormwater Management and Urban Renewal Authority decided this was an ideal time to implement the 20-year-old redevelopment plan (WWE and Flanagan, 1986). The damaged homes were acquired by the City and the residential pocket cleared. Stormwater Management began working on a plan to correct the flooding problem so the area could be redeveloped. In addition to drainage channels and storm sewers, a stormwater detention pond was needed to store water during periods of heavy rainfall.

The site is located in the alluvial flood plain of the Arkansas River, with a high water table. Further, the oil refineries had been polluting the soils for decades with petroleum products, and urban development was encroaching on the few remaining habitats of the Least Tern, an endangered species.

A creative engineering consultant, Wright Water Engineers of Denver, after careful analysis of the constraints and needs of all parties involved, developed a unique multi-purpose pond that solved the multitude of problems faced (WWE, 1987).

The solution to the problem had to be very cost effective, if not downright cheap, due to tight budgetary constraints. The sandy soils and high water table required a pump station to be installed for a dry detention pond. Further, a dry detention pond would continually leach petroleum pollution from the saturated soils into a pond, which would be pumped into the Arkansas River.

The solution was a wet detention pond that took advantage of the high water table and prevented leaching of oils. (See figure 2.) The Nature Conservancy was consulted, and the pond was designed to provide a habitat for the Least Tern. An island was constructed in the center to provide safe haven from domestic pets. A trail was designed around the perimeter to provide maintenance access as well as hiking and nature observation. The pond is to be stocked with

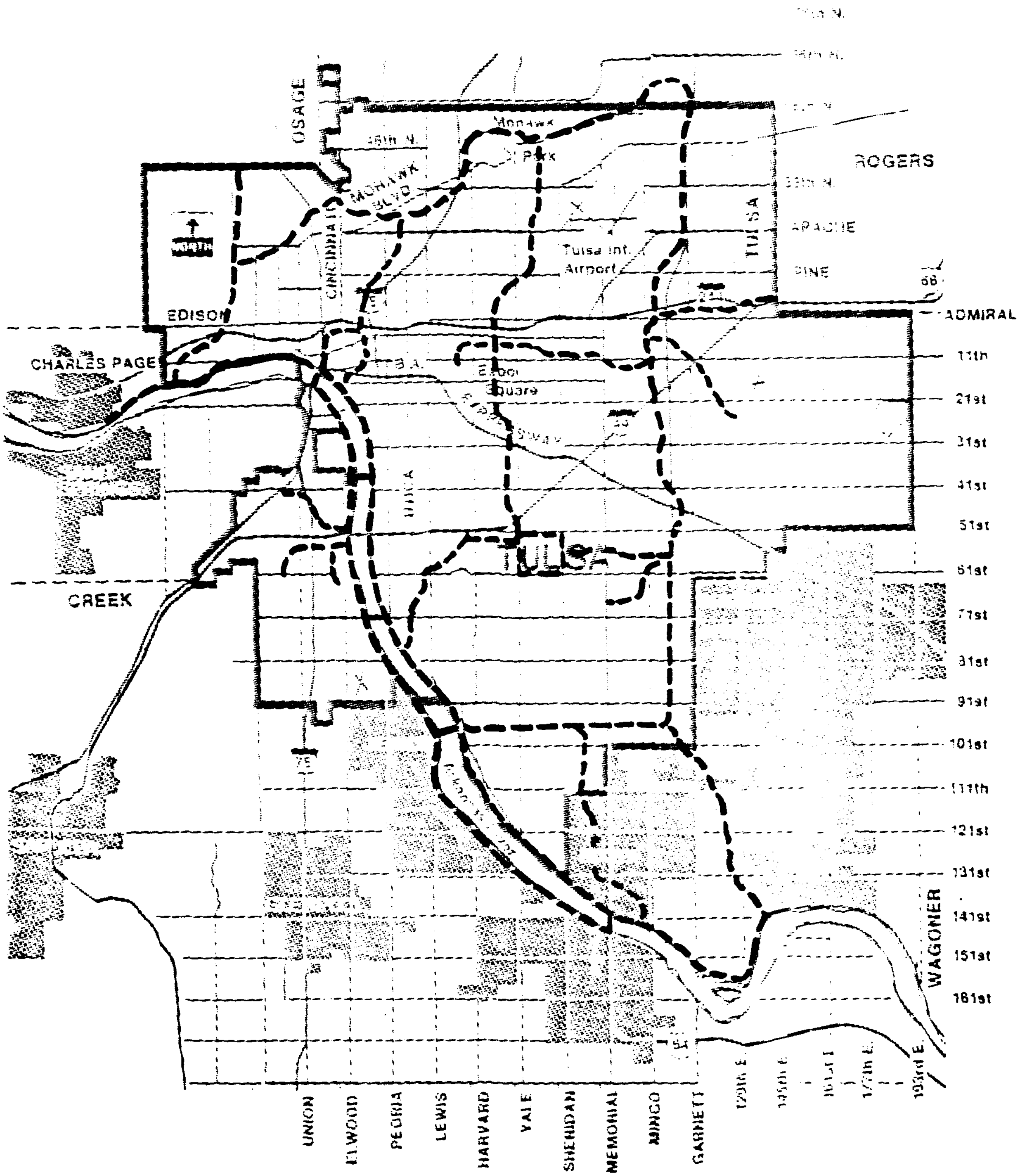

FIGURE 1. TULSA TRAILS CONCEPTUAL PLAN

FIGURE 2. GARDEN CITY MULTI-PURPOSE STORMWATER DETENTION POND

Gambusia to control mosquito populations, and the natural character of the pond provides visual relief from the surrounding industrial environment. Through this one project, the objectives of many agencies were accomplished.

AMERICANS OUTDOORS: THE KEY TO SUCCESS

President Reagan appointed a blue-ribbon commission to develop an alternative to the Land and Water Conservation Fund, which now has $11 billion in its account. This President's Commission on American Outdoors has recommended spending $1 billion a year to create trail systems and greenway corridors throughout the nation (Americans Outdoors, 1987). It specifically mentions preservation of floodplains and wetlands, abatement of air and water pollution, preservation of fish and wildlife habitat and vegetation stands.

Every environmentalist and recreationist needs to support this recommendation, for within it are the keys to the success of all our programs.

ENVIRONMENTAL COALITION: STRATEGY FOR THE FUTURE

The central theme of these case studies is that we need to think "multi-purpose use." We need to form a national coalition of environmental and recreation groups to accomplish these cherished objectives.

The land use area with which we are concerned comprises about 10% of the total land area of the nation. We need to acquire that ten percent--primarily floodplains--and place it in the public sector for future generations.

This sounds like a large-order public acquisition of the nation's floodplains. But if all the monies of all groups were pooled, the entire area could be preserved in just a few years. A billion dollars a year from the Land and Water Conservation fund would go a long way toward that objective (just 20% per year of the nation's flood damages would save $800 million). From a cooperative perspective, we can accomplish more together than we could ever envision alone.

ENVIRONMENTAL BASE MAPPING: THE FIRST STEP

The first step in any planning study is an inventory. With high-speed, computer-aided CAD/ GIS mapping systems, we could create and map a data base of key significant environmental areas.

An inventory, on a state-by-state basis, in layers for computer mapping could be provided. Floodplains, wetlands, wildlife habitat, vegetation, scenic corridors, scenic rivers, trail locations, etc., could be layered, and by layering, thus prioritized. Value systems could be added--for example, threat of impending loss to urbanization, endangered species habitat, etc.--and priorities for acquisition ranked.

At present the most pressing impediment to this step is the lack of a central authority, such as the U.S. Geological Survey, to assume responsibility for the development, operation and management of such a system into which groups and states could feed their data.

CONCLUSION

To conclude, we must begin to think in terms of multi-purpose use to gain the biggest bang for the buck. Each public dollar must accomplish as many public objectives as possible. We must expand our support base to include recreation. The President's Commission on Americans Outdoors has recommended a program to preserve our nation's valuable waterside lands. We must, however, take the next logical step in its implementation - we must acquire the critical areas: the drainageway corridors and floodplains. We must begin with a nationwide inventory and develop a computer-based mapping system to aid in location and prioritization of many individual layers of data. And finally, we must all pool our funds, our creative efforts, and our energies to make it happen. This paper is a Call to Action.

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Acquisition and the Federal Emergency Management Agency Flood Loss Reduction Program

Frank H. Thomas
Federal Emergency Management Agency

INTRODUCTION

The subject of multi-objective greenways for floodplains and wetlands can be addressed in the context of a very important national problem: the mitigation of risk from natural hazards. Natural hazards (flooding, erosion, soil compaction, liquefaction) and wetlands enjoy a frequent geographic coincidence which presents opportunities for collaborative hazard and wetlands management. This coincidence is especially true for flood hazard which represents the most frequent cause of natural hazard losses. The following comments describe the nature of the natural hazard problem, discuss experience with floodplain management and especially the acquisition program of the National Flood Insurance Program, and suggest opportunities for collaborative wetlands/floodplain management.

NATIONAL HAZARDS AS A NATIONAL PROBLEM

The risk associated with some natural hazards is determined by geographic site characteristics, e.g., the depth of flooding in floodplains, the intensity of shaking in seismic zones, and the angle of slope in landslide areas. Among these hazards, flooding is the most frequently encountered and is consistently responsible for a large amount of average annual losses. Moreover, most of the nation's wetlands are located within riverine and coastal floodplains.

A concept of the magnitude of the flood hazard problem can be obtained by looking at some characteristics of the floodplain, which we define as the area subject to a one percent chance of flooding in any given year. Two different data bases indicate the size of the floodplain area. National Resource Inventory data from the U.S. Department of Agriculture indicate that outside of Federal lands and urban areas, 304,344 square miles or 10 percent of the rural land is in floodplain (U.S. Department of Agriculture, 1982). National Flood Insurance Program data of the Federal Emergency Management Agency (FEMA) indicate that there are 253,850 square miles of floodplain in the 17,750 communities participating the Program (Federal Emergency Management Agency, 1983). The floodplain of these communities provides residence for an estimated 45.3 million people. In terms of average annual losses, flooding costs the Nation 163 (U.S. Department of Commerce, 1987) lives and between $3-4 billion (U.S. Water Resources Council, 1975). The seriousness of the national flood problem has been recognized repeatedly by the Congress which since the 1930's has enacted legislation creating more than 25 programs to reduce flood losses (Federal Emergency Management Agency, 1986). Assignment of authority for these programs to different agencies has resulted in problems of coordination. To cope with these problems, the Congress directed the President to develop and report to the Congress on a Unified National Program to reduce flood losses (The Congress, 1968). In addition, the President issued an Executive Order directing Federal agencies to avoid actions adversely affecting floodplains (The President, 1977). Unfortunately, much of the fragmentation of program responsibility found at the Federal level is also characteristic of State and local government.

Clearly, flooding is a major national problem and although experience with flood hazard is greatest, other natural hazards such as earthquakes or landslides with less well defined high hazard areas may also be coincident with wetlands and offer multiobjective management opportunities.

FLOOD LOSS REDUCTION EXPERIENCE

In 1965 when the Congress established the Water Resources Council as a federal interagency body, it was greatly concerned with coordination among Federal agency programs and the need for multi-objective water resources planning. It regarded flood control as one objective (The Congress, 1965). In 1968 the President assigned responsibility for developing a Unified National Program for Floodplain Management to the Water Resources Council. This Unified National Program reflects an interagency approach sensitive to the competing national objectives of economic development and environmental quality.

The Unified Program sets forth a conceptual framework for management in which the goal of floodplain management is the wise use of floodplains, or, uses compatible with the risk to human life and property and the risk to natural and beneficial functions served by the floodplain. The Unified Program identifies three loss

reduction strategies (modify the flood waters, modify the susceptibility to flooding, and modify the impact of flooding) and a set of tools (e.g., flood control structures, property acquisition, building codes and zoning ordinances) for implementing each strategy. Similarly, it identifies two strategies for reducing losses to natural values (restoration and preservation), and a set of tools (regulations, development policies, tax adjustments, etc.). It argues that for any given set of floodplain conditions and the values of the local community, the best mix of loss reduction strategies and tools should be sought. Thus, the Unified Program does not prejudge any one floodplain use (objective) or any one loss reduction tool as superior while at the same time it recognizes that the authority for objectives and loss reduction tools is partitioned among levels of government and the private sector. Finally, the Unified Program recommends steps to be taken to achieve improved implementation of floodplain management.

Since it was first adopted in 1976 and through revisions in 1979 and 1986, the Unified National Program has served as a focal point for Federal agencies with different and sometimes competing program responsibilities. It has permitted them to work together toward achievement of more effective management of the Nation's floodplains. Under the auspices of the Program, in 1977 a series of five workshops was held on integrated wetland-floodplain management. Since then one or more workshops have been held each year, focusing on opportunities for interagency and to some degree, intergovernmental coordination of programs. Currently, through joint funding by seven Federal agencies, a national status report and an evaluation of effectiveness is being prepared for each of the loss reduction tools identified in the Unified Program.

Experience and success with the Unified Program suggests an approach for advancing floodplain as well as wetland management which must operate in a milieu of competing economic development and environmental quality objectives with program authority divided within and among the levels of government.

ACQUISITION AND NATIONAL FLOOD INSURANCE PROGRAM

The objective of the National Flood Insurance Program (NFIP) is to reduce future disaster assistance and flood insurance claim payments. This objective is served by two basic strategies: protecting structures from flood waters by use of elevation techniques and flood resistant design and materials, and, keeping structures away from floodprone areas by avoidance and removal. The removal of insured structures that have been severely or repeatedly flood damaged may be accomplished by the Section 1362 flood damaged property purchase program of the NFIP. This program enables owners of insured flood damaged structures to be permanently removed from flood risk areas and it supports local and other governmental units in the management of flood risk areas. In eight years of program operation, projects in 60 communities have resulted in the purchase and removal of 992 structures with an expenditure of $37 million.

To be eligible for a property purchase project, a community participating in the NFIP must adopt and commit to carry out a satisfactory reuse plan for the property to be acquired and also agree to remove the purchased structures and maintain the land as open space. To be eligible, an individual property owner must be located in a flood hazard area, own the structure and the land, be insured at the time of damage, and have a structure which meets one of the following criteria: (1) substantially damaged beyond repair; (2) flooded three times in five years sustaining damage on each occasion equal to 25% of the structure's value, or (3) repair prohibited by statute, ordinance or regulation. The purchase price is determined by subtracting the amount of the flood insurance claim paid from the pre-flood fair market value of the structure and land and offering the difference to the property owner. The decision of the property owner to sell is voluntary.

Experience with the Section 1362 Program reveals four attributes of the most successful projects. First, the most cost effective projects have 15 or more eligible properties. Second, the time elapsed between the flood event and the actual property purchase is minimized, certainly less than 12 months. Third, projects wherein the community participates on a cost sharing basis are more cost effective and result in purchase of a larger number of properties. Fourth, the larger the number of geographically contiguous properties purchased, the more successful the reuse plan is likely to be.

Several program eligibility requirements have an important impact on whether the land purchased results in a contiguous or checkerboard pattern of public ownership. First, not all flood damaged property is insured. Second, flooding may damage structures differently and not all structures in an area may meet the severe or repetitive damage criteria. Third, voluntary sale may result in some property owners choosing not to sell; the ratio of purchase to eligible properties is 70 percent. Consequently most projects do result in public ownership of noncontiguous properties and the local community must seek to acquire property through other programs and/or suffer a difficult property maintenance problem.

In addition, it must be noted that each project has its own unique problems. This can be demonstrated by looking at Baytown, Texas;

Tulsa, Oklahoma; Hamburg, New York, and the four adjacent New Jersey communities of Wayne, Oakland, Fairfield and Lincoln Park.

Baytown, Texas. The Brownwood Subdivision of Baytown, Texas was a middle income coastal subdivision of approximately 300 homes where 9 feet of land subsidence had placed most structures 0-2 feet above mean high tide and 14-16 feet below the base flood elevation. Frequent flooding has caused millions of dollars in damages including an estimated $3 million in NFIP claims and a 50 percent or more depreciation of fair market value. In 1983, Hurricane Alicia inundated the whole area causing an estimated $6 million in NFIP claims, $1.5 million for temporary housing and individual assistance, and another $1.5 million damage to sewers, waterlines and other public facilities.

After insurance claims had been paid, a Section 1362 project acquired 177 structures for a very modest $551,000 or an average of $3,100 per property, a significant direct cost savings when the frequency of expected flooding is considered. The city has not repaired the damaged public facilities and is using the money saved for redevelopment of the area. Moreover, the city of Baytown is in the process of acquiring and clearing the remaining properties and within two years is expected to begin improving the area for a public park and wildlife refuge. The result of this project will be a single tract of more than 125 acres of contiguous coastal open space plus the elimination of repetitive flood disaster costs.

Tulsa, Oklahoma. The City of Tulsa, Oklahoma experienced two major riverine floods in May, 1984 immediately following a highly contested mayoral election in which flood loss reduction was a major campaign issue. The Section 1362 project was initiated with the strong commitment of a new mayor and with an exceptionally high degree of public awareness. Of 96 eligible properties, 89 were purchased at a cost of $1.9 million or an average $21,000 per parcel. The entire Section 1362 effort was completed less than one year after the flood event. The city adopted and pursued a comprehensive post-flood mitigation program which included the purchase of additional parcels of land and the expenditure of funds exceeding the FEMA Section 1362 expenditures. The result of this project is the creation of a corridor of riverine open space and the elimination of significant repetitive flood losses borne by both the public and individual property owners.

Hamburg, New York. In 1985, the Town of Hamburg, New York experienced serious flood damage to 23 lakefront properties and qualified for a Section 1362 project. This project has been delayed and complicated by separation of the ownership of the individual structures from the land which is held by a homeowners association. An otherwise highly qualified project of contiguous lakefront properties suitable for open space use is in jeopardy because of legal-ownership issues.

Wayne, New Jersey. The cities of Wayne, Oakland, Fairfield and Lincoln Park, New Jersey have riverine areas which have experienced repeated flooding involving over 100 eligible properties. Since the time of flooding, reduced gasoline prices have made these communities attractive for commuters to the New York City area. As a result, property values have been rapidly appreciating in spite of the known flood history, and most property owners are no longer interested in participating in the Section 1362 program.

SUMMARY

From the brief survey of natural hazards, the flood loss reduction experience, and the NFIP acquisition program, generalizations can be drawn regarding multi-objective greenways for floodplains and wetlands:

(1) Site specific severe natural hazard areas may be highly suitable for greenway use; this is certainly true for coastal and riverine high hazard areas.

(2) Authority and programs for dealing with flood loss reduction are fragmented across numerous Federal agencies, a situation also found at the state and local levels.

(3) A National coordinating mechanism, A Unified National Program For Floodplain Management, has proven to be a valuable vehicle for conceptualizing flood loss reduction activities and identifying program status and needs.

(4) The flood damage property acquisition program of the NFIP is a relatively small program that can successfully be articulated with greenway objectives but usually needs to be supplemented with other funding resources if contiguous property purchase is to be achieved.

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The Implications of Hydrology and Landscape Ecology on the Maintenance of Freshwater Wetland Ecosystems in Florida

Kevin L. Erwin
Kevin L. Erwin Consulting Ecologist, Inc.

INTRODUCTION

Floridians have only recently realized that surface water management is a major concern of local, regional and national significance. Water cannot be considered a free or inexpensive resource to be rapidly drained from the land in preparation for development. Since the early 1970's, regulatory agencies, local governments, and environmental organizations have expended great effort in protecting wetland habitats. Unfortunately, at the same time surface water drainage practices have continued. Hydrology and landscape ecology must be recognized as critical factors by wetland managers as they strive to attain the goals of managing, preserving, creating, or enhancing wetland systems and their functions and values.

Over the last 200 years, 30 to 50% of the wetlands of the continental United States have been destroyed by such diverse uses as agriculture, mining, forestry, and urbanization (U.S. Congress, 1984). Since the 1970's, government regulation of wetlands in Florida has slowed the destruction of this habitat which has decreased by 12,000,000 acres between 1850 and 1973, a 60% loss (Tschinkel, 1984). Based on satellite imagery taken between 1972 and 1974, Hampson (1984) estimated the total wetland acreage in Florida at 8.4 million acres, a decrease of 13.4 million since 1955. This figure includes the loss of tidal estuarine wetlands such as salt marshes and mangrove swamps. However, a significant percentage of wetlands which have been destroyed also consisted of interior freshwater wetlands. Cypress strands, domes and forests, river swamps and flood plains, other forested wetlands such as bay heads, gum swamps and hydric hammocks, and marshes ranging from the small isolated variety to large contiguous systems such as the Everglades have all been impacted.

While the major emphasis of surface water management during the last decade has been on water quality enhancement and conservation, the major focus of wetlands protection has been on habitat preservation. Since 1973, a number of State and Federal laws have been passed which provide a great measure of protection to wetlands through preservation or conservation efforts by requiring permit application review for activities proposed within wetlands. This legislation (e.g. Section 404 of the Federal Clean Water Act dredge and fill permitting program and the State of Florida's 1984 Henderson Wetlands Protection Act) contains rigid guidelines for the identification of wetlands jurisdiction and mandates the avoidance of wetlands by proposed developments. Another emphasis on wetlands protection has been the acquisition of wetlands through beneficial state programs such as the Conservation and Recreation Lands Program managed by the Florida Department of Natural Resources and the Save Our Rivers Program managed by the five Water Management Districts. On large tracts of land some comprehensive measure of attention is directed towards the management of the on-site wetlands either through acquisition or as a result of the Development of Regional Impact (DRI) review process (Chapter 380 Florida Statutes). However, over 90% of the total number of land development projects within the state are not DRI's and rarely are wetlands in any type of permitted project including agriculture, acquired and/or systematically managed.

The identification and preservation of wetlands by the permitting process will not insure the long term viability of these habitats or guarantee their function as productive ecosystems which provide fish and wildlife habitat, water storage, and recreational areas. If they are not evaluated and managed as a complete natural system, such wetlands can be expected to exhibit significantly lower faunal and floral species composition and diversity as a result of altered water level and hydroperiod, coupled with isolation from natural upland habitats. The greatest short-coming of our efforts to protect wetlands is the current dichotomy which exists between wetlands regulation and both surface water management and adjacent land use planning. While preserving wetlands surrounded by intense urban and agricultural land uses, we may be failing to incorporate these habitats into management units which combine proper hydrology and upland buffers. Should we continue this practice, the wetlands preserved through permitting will diminish in value over time.

ECOLOGICAL RATIONALE

When a wetland is preserved, the long term goal of maintaining all of its vital functions at optimal levels should be of primary concern. Hydrology is the <u>single</u> most important element

required for the maintenance of specific types of wetland ecosystems and their functions. Hydrologic conditions can directly modify or change chemical or physical properties such as nutrient availability, degree of substrate anoxia, sediment properties, and pH. Water inputs are invariably the dominant source of nutrients to wetlands; water outflows often remove biotic and abiotic material from wetlands (Mitsch and Gosselink, 1986). These modifications of the physio-chemical environment, in turn, have a direct impact on the biotic response in the wetland (Gosselink and Turner, 1978). Wetland fauna and flora will often respond to even slight changes in hydrologic conditions with resulting substantial changes in species richness, diversity and productivity. Thus, an abrupt and usually significant change in the functional integrity of the wetland system often results from a minor change in its hydrology.

In southern Florida drainage and diversion of water has greatly reduced the amount of water held in the region and has intensified and prolonged the normal winter dry season. This, in turn, has greatly increased the number and destructiveness of fires (Craighead, 1971). In addition, abnormally high water levels in the Everglades water conservation areas have resulted in dramatic adverse impacts to a wetland system of national significance. Everglades habitats have been significantly altered and wildlife biologists believe the artificially increased water levels and extended hydroperiod have been a significant factor in the 90% reduction of wading bird populations that has occurred.

One of the greatest threats to hydrologically altered wetlands in south and central Florida is the rapid conversion of these wetlands to areas completely dominated by the problematic exotics Melaleuca quinquenervia and Schinus terebinthifolius. These trees rapidly invade wetlands with altered hydroperiods. In a period of perhaps less than 20 years a monoculture of these species and, except for continued water storage functions, a significant reduction of wetland functions could result. Once established, these exotics can tolerate flooding extremely well. Therefore, restoring hydroperiod alone to hydrologically altered wetlands with exotic infestation is not a method of eradication. Physical removal with regular follow-ups required is often the only solution.

Exotic infestation is one of the single greatest threats to the long term viability of all freshwater wetlands in south Florida. There is no reliable data which describes the acreage and location of hydrologically altered wetlands or wetlands currently infested with Melaleuca and/or Schinus. However, large systems such as the Everglades National Park, Big Cypress National Preserve and Six Mile Cypress Strand contain significant infestations where management/eradication will require substantial financial commitment. Most of the isolated marshes and cypress domes in urbanized south Florida which have been regulated show signs of exotic infestation.

MULTI-OBJECTIVE GREENWAYS CONCEPT - INCORPORATING WETLANDS AND UPLAND LANDSCAPES

There is a need in Florida and throughout the nation to evaluate wetlands as whole ecosystems which need to be connected by a variety of habitats and hydrological pathways. Establishment of a multi-objective greenway strategy should be used to insure that wetlands that are preserved or created through mitigation will be able to survive future natural and man-made perturbations within the watershed and function at optimal levels. The objective of a greenway is to integrate wetlands and uplands along a contiguous hydroecological pathway conducive to the maintenance and perpetuation of the system, irrespective of property or political boundaries. Wetlands properly connected with native or restored upland habitat will function as wildlife corridors and provide natural low energy surface water management via storage and sheetflow instead of ditches, dikes and pumps.

Since hydrology is the single most important element to be considered for the optimal maintenance of wetland ecosystems, proper hydrologic analysis is imperative. The basic parameters to consider include: identification of the watershed and sub-basin; computation of inflows and outflows from the property and each wetland; a comparison of the pre- and post-development flow characteristics identified above on a seasonal basis; the average water depth; the maximum flooding depth; the maximum dry-down depth; the hydroperiod or duration of flooding and the date of normal wet-season flooding. These measurements should be taken at a standard point at the upland edge of the wetland over at least one typical water year and including two wet seasons. The collection of the hydrological data and proper ecological evaluation will define the boundaries and design of the greenway. Needs of particular wildlife species relate directly to management objectives and will influence the design of the greenway. It is obvious that the successful use of this approach requires the evaluation of the system as a whole rather than isolated areas limited by ownership or political boundaries as is currently the practice with most current dredge and fill or water management permitting.

The current regulatory procedure of evaluation on a case-by-case, parcel-by-parcel basis properly identifies the wetland boundaries on each property and provides some conservation of the wetland and the mitigation of disrupted habitat. However, the end result is often undeveloped wetland habitat completely surrounded by intense urban or agricultural development. The hydrologic system and landscape ecology is

disrupted and the upland-wetland ecotone eliminated. Habitat fragmentation is a serious threat to biological diversity and is a primary cause of the current extinction crisis (Wilcox and Murphy 1985). With the absence of interconnecting wildlife corridors, fish and wildlife values are significantly diminished. Noss (1987) provides a comprehensive and thorough discussion of the potential advantages of corridors to conservation of terrestrial species and habitats in human-dominated landscapes that are not discussed here.

An alternative to this scenario is the systematic approach to wetlands management requiring the implementation of multi-objective greenways. Under this approach a development permit application for a parcel of land would be evaluated as part of a larger system (watershed). The parcel and proposed development would be properly designed to maintain its pro rata contribution to the normal or desired carrying capacity and resource related functions of the watershed. This process of evaluating watersheds obviously requires a significant expenditure of effort, but it would insure much improved, data based land use decisions. An opportunity for this type of procedure currently exists. The requirements of Florida's 1985 Local Government Comprehensive Planning and Land Development Regulation Act (particularly Section 1633178, Florida Statutes) and the 1986 Florida Department of Community Affairs Minimum Criteria for Review of Local Government Comprehensive Plans and Determination of Compliance (Chapter 9J-5, Florida Administrative Code) can facilitate comprehensive data collection on a regional-watershed basis for each municipality. This approach is currently being utilized in Lee and Volusia Counties' Coastal Studies (Erwin, 1988 and K.L. Erwin Consulting Ecologist, Inc., 1988) and should be considered throughout the county. Since the data collected from these efforts would be invaluable to a number of local state and federal agencies an opportunity exists for cost sharing.

The hydrological approach to wetland evaluation and permitting will improve wetland managers' ability to retain functioning, maintainable ecosystems. The benefits to the general public and to the landowner are obvious with less money spent on exotic control, drainage, and improved recreational opportunities. Land use decisions based upon scientific data are usually positively received by environmentalists, landowners, developers, and the judicial system which often must review appeals of regulatory decisions. One major benefit however, is that those wetlands protected will actually continue to function as a larger ecosystem in the future. A balance of wetland and required upland conservation may result from utilizing greenways, requiring some compromise on current trends towards near total wetland preservation and high upland utilization. However, the end result would be the retention of interconnected, functioning, maintainable "hydro-ecological" ecosystems where the remaining wetland and uplands function as a productive unit resembling the original landscape.

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chapter twelve

Research Needs

WETLAND MANAGEMENT: HYDRAULIC AND HYDROLOGIC RESEARCH NEEDS

Although additional highly focused research is needed, the first priority for wetland managers may be better access to and use of existing hydrologic research and hydrologic data. Guidebooks, training exercises, workshops and other approaches are needed to disseminate information and train managers.

In the last decade, efforts to actively manage water regimes in wetlands and to restore and create wetland systems have focused and clarified "applied" hydrologic research needs. Such needs relate not only to the ways in which wetlands function under various hydrologic conditions but to the assessment of specific functions in particular contexts, including the sensitivity of functions to various types of impacts and ways in which various functions can be restored or recreated.

A variety of wetland hydrologic research needs were set forth in a theme paper concerning "Water Resources and Wetlands" prepared by Carter, Bedinger, Novitzki and Wilen for Greeson, Clark and Clark (ed.), *Wetland Functions and Values: the State of Our Understanding*, American Water Resources Association (1978). These needs are equally applicable today although progress has been made with regard to several topics and a variety of additional, more specific needs may be added. Some of the overall categories of research considered "most urgent" by the 1978 paper and of equal relevance today include:

--"Improving, refining and perhaps simplifying existing techniques for hydrologic measurements." Low cost approaches for continuous monitoring of surface and ground water levels and water quality are particularly needed.

--"Making accurate measurements of all the hydrologic inputs and outputs to representative wetland types and estimating the errors inherent in various measurement techniques." Measures of groundwater recharge and discharge to ephemeral wetlands and wetlands with long fluctuations in water levels are a particularly high priority. Estimation of errors inherent in not only measurement but modeling is also needed.

--Improving "our basic understanding of...the soil-water-vegetation relationships of wetlands." Understanding of short term and long term wetland/mineral sediment relationships as well as organic substrate relationships is much needed.

--Making "in depth, long-term studies of different wetland types under different environmental conditions." Hydrologic measurement of maxima, minima, hydroperiod, and "mean" as related to particular vegetation, fauna and wetland functions would be a component of such studies.

--Developing "models based on hydrologic data so that we have better analysis and predictive capability." This effort should include grouping or "classification" of wetlands based upon hydrological and geomorphological characteristics and the tailoring of models to the various classes. Models should address both small and large scale events.

To this list may be added several additional "urgent" research needs. Many of these may be considered subsets of the above:

--Hydrology and geomorphology. Improved understanding of the relationship of hydrological to geomorphological processes is needed including the relevance of particular hydrologic events and their characteristics (e.g., size, frequency) to the location, shape, size, depth, soil regime and other features of wetlands. The time frame for such changes should be studied. This knowledge is essential not only for understanding wetland functions and restoration/creation needs, but for predicting the impact of hydrologic modifications caused by activities such as dikes, dams, levees, drainage, and channelization, and for efforts to manage, restore and create wetlands. Geomorphologists are well as hydrologists should be involved in such research.

--Hydrologic maxima and minima, norms. Improved understanding is needed of the role of hydrologic and hydraulic maxima and minimum, mean conditions, and hydroperiod with regard to particular plant and animal species. This knowledge is needed to determine the sensitivity of particular wetlands to changes in hydrologic regimes, to predict changes, to determine the role of extreme events in wetland plant "succession", and for all aspects of wetland water manipulation, restoration, and creation. Such research should be combined efforts of hydrologists and wetland botanists and biologists.

--Long term fluctuations. Improved understanding of the role of long term fluctuations in water levels due to climatic cycles upon the location, form, and other characteristics

of wetlands is needed. This information is needed to determine long term management needs including buffers for wetlands. Climatologists, hydrologists, geomorphologists, botanists, and biologists should be involved in such research.

--**Wetland/sediment relationships.** Improved understanding of wetland/sediment relationships including both the negative and positive roles of sedimentation. This knowledge is needed to assess and predict the impact of watershed erosion and sediment upon wetland functions such as habitat, nutrient supply, and pollution uptake; the impact of reservoirs and other activities that affect sediment supply; and the impact of sea level rise on coastal and riverine wetlands and the ability of wetlands to respond to such rises. It is also needed to design restoration and creation projects and to predict the probable "life" of created systems. Geomorphologists and soil scientists as well as hydrologists and more traditional wetland researchers should be involved in such research.

--**Impacts of activities upon hydrology.** Improved understanding is needed of the short and long term impacts of various activities upon wetland hydrology, the resulting changes in wetland functions, and techniques for reducing those impacts. The long and short term impacts of fills, agriculture, forestry, reservoirs and other activities should be evaluated and compared.

--**Wetland grouping of classifications based upon hydrologic or hydrologic/geomorphological characteristics.** Various ways should be developed for grouping wetlands based upon hydrologic or hydrologic/geomorphological characteristics to aid in the assessment of wetland functions, sensitivity to impacts, and potential for restoration or creation. Approaches already suggested by Zimmerman, Novitzki, and O'Brien should be examined for application in particular contexts. Geomorphologists as well as hydrologists and more traditional wetland scientists should be involved.

--**Relationship of hydrology to particular wetland functions.** Our understanding of the importance of hydrology to various wetland functions including the assessment of such functions is needed. This research should consider the impacts of various hydrologic modifications upon those functions, and approaches for restoring, enhancing, or creating particular functions:

1. Flood conveyance. This function of riverine wetlands has been largely ignored (no known papers) by wetland managers although strongly emphasized by floodplain managers and the subject of considerable modeling and mapping. For a start, riverine wetland maps and FEMA floodway maps could be compared for sample areas.

2. Flood conveyance. An effort should be made to synthesize and explain differences in various empirical studies of wetland flood storage and provide an analytical model for predicting storage in particular contexts. The role of storage for various magnitudes of flood events should be further explored. The role of the topography (bank, terraces) immediately surrounding a wetland should also be clarified since it appears that many wetlands will function as reservoirs only if the surrounding bank or rim (not technically part of the wetland for scientific or regulatory purposes) is kept intact.

3. Groundwater recharge and discharge. Although there is evidence that many wetlands are discharge areas at least a portion of the year, the recharge role of ephemeral wetlands (dry part of the year) should be investigated as well as the recharge role of other wetlands during periods of high water. The role of discharge into wetlands should also be investigated including the effects of discharge upon wetland soil development, vegetation, and fauna.

4. Erosion and sediment control. Existing studies concerning the functions of various wetland and riparian zone species (e.g., cottonwood) for bank and bed stabilization, control of stream meander, wave attenuation, and sediment retention should be continued. Additional research is needed which focuses upon erosion and sediment during large flood events.

5. Water quality protection. Additional hydrologic or hydrologically-related research is needed on many aspects of wetland water quality protection functions:

   -- Erosion and sediment control. See item (4) above.

   -- The role of riparian soils and vegetation as "buffers" for lakes, streams, estuaries, wetlands. Additional empirical studies similar to those carried out by J.W. Gilliam and his colleagues are needed which address the short and long term effectiveness of various wetland and riparian zone vegetation and soil types in intercepting sediment, nutrients, toxic metals, pesticides, and other pollutants before they reach lakes, streams, estuaries, and other water bodies.

   -- The role of wetlands in trapping, transporting, and (perhaps) ultimately

releasing pollutants. Additional studies are needed with regard to the role wetland vegetation, fauna, and soils may play in trapping and transforming various pollutants within wetlands and the short term and long term impacts of these pollutants upon flora and fauna. Additional studies are also urgently needed with regard to potential long term movement of pollutants out of wetland systems through sediment transport, flood flows, normal flows, flood chain, and other mechanisms.

6. Habitat (fisheries, waterfowl, mammals, reptiles). Many additional species-specific and ecosystem-specific studies are needed with regard to the role of various hydrologic maxima, minima, mean conditions upon vegetation, soils, and the broad range of breeding, feeding, resting and other habitat needs of particular species.

7. Food chain support. A broad range of additional studies is needed to understand the relationship between various maxima, minima, and mean hydrologic conditions and various levels of productivity, the transformation of food chain support materials in various wetland settings, and the transport of such materials to and from wetlands under various conditions.

8. Recreational values. Additional empirical research and synthesis of existing information is needed to relate various hydrologic maxima and minima including water depth, velocity, etc. to particular recreational uses in wetlands including canoeing, birdwatching, boating, hunting, etc.

To reduce costs, wetland researchers wishing to carry out hydrologic studies might incorporate such studies into wetland creation or restoration projects. Considerable control may also be provided in such contexts. Post flood studies after naturally occurring flood events could also provide invaluable information concerning the response of soils, flora, and fauna to various "extreme" conditions.

GLOSSARY OF TECHNICAL TERMS

AQUIFER	A body of rock or soil that contains sufficient saturated permeable material to conduct groundwater and to yield economically significant quantities of groundwater to springs or wells.
ARTESIAN	Refers to water confined under pressure.
BENCH MARK	A fixed, more or less permanent reference point or object, the elevation of which is known. The U.S. Geological Survey installs brass caps in bridge abutments or otherwise permanently sets bench marks at convenient locations nationwide. The elevations on these marks are referenced to the National Geodetic Vertical Datum (NGVD), also commonly kown as Mean Sea Level (MSL). Locations of these bench marks on USGS quadrangle maps are shown as small triangles.
CHANNEL GEOMETRY	Description of the shape of a given cross-section of a river channel.
CONFINED AQUIFER	An aquifer bounded above and below by beds of material distinctly less permeable than the aquifer itself.
CONTOUR	An imaginary line of constant elevation on the ground. The corresponding line on a map is called the contour line.
DISCHARGE (STREAM)	Rate of flow of a stream expressed as volume per unit of time.
ENHANCEMENT	An activity of man which increases one or more natural or artificial wetland functions.
EUTROPHICATION	The process by which bodies of water become enriched with mineral nutrients and organic materials, often causing increased algae growth.
FLOOD	A temporary rise in stream level that results in inundation of areas not ordinarily covered by water.
FLOODWAY	The channel of the watercourse and those portions of the adjoining flood plains which are reasonably required to carry and discharge the regulatory flood.
FLOOD FREQUENCY	The average frequency, statistically determined, for which it is expected that a specific flood level or discharge may be equalled or exceeded.
HYDRIC SOIL	A soil that is saturated, flooded, or ponded long enough during the growing season to develop anaerobic conditions that favor the growth and regeneration of hydrophytic vegetation.
HYDROLOGY	The science dealing with the properties, distribution, and circulation of water.
HYDROPHYTE	Any macrophyte that grows in water or on a substrate that is at least periodically deficient in oxygen as a result of excessive water content; plants typically found in wetland habitats.
INUNDATION	A condition in which water from any source temporarily or permanently covers a land surface.
MEAN SEA LEVEL	A datum, or "plane of zero elevation", established by averaging all stages of oceanic tides over a 19-year tidal cycle or "epoch". This plane is corrected for curvature of the earth and is the standard reference for elevations on the earth's surface. The correct term for mean sea level is the National Geodetic Vertical Datum (NGVD).

NATURAL	Occurring within a physical system which has developed without human intervention, in which natural processes continue to take place.
PERMEABILITY	The property of soil or rock to transmit water or air.
PERSISTENCE	The overall ability of a wetland to continue to exist as a wetland and to serve wetland functions over a period of time although its vegetation, soils, hydrologic chararacteristics, and precise boundaries may change.
PIEZOMETRIC SURFACE	The level to which water from an aquifer will rise in a well. Also called the Potentiometric Surface.
POORLY DRAINED	Soils that are commonly wet at or near the surface during a sufficient part of the year that field crops cannot be grown under natural conditions. Poorly drained conditions are caused by a saturated zone, a layer with low hydraulic conductivity, seepage, or a combination of these conditions.
REACH	A hydraulic engineering term to describe a longitudinal segment of a stream or river.
RECHARGE AREA	An area in which water is absorbed and added to the groundwater system.
SCOUR	The clearing and digging action of flowing water, especially the downward erosion by stream water in sweeping away mud and silt on the outside of a curve or during a flood.
SEDIMENT LOAD	The quantity of solid material that is transported by a natural agent, such as a stream, and expressed as dry weight passing a given point in a given period of time.
SUCCESSION	The progression or gradual replacement of one kind of biotic community by another, the compositions of which overlap in the process.
TOPOGRAPHY	The configuration of a surface, including its relief and the position of its natural and man-made features.
TRANSITION ZONE	As used here, the area in which a change from wetlands to upland occurs. The transition zone may be narrow or broad.
UNCONSOLIDATED	With reference to earth materials, materials whose particles are not cemented together and are loosely arranged.
UPLAND	As used here, any area that does not qualify as a wetland because the associated hydrologic regime is not sufficiently wet to elicit development of vegetation, soils, and/or hydrologic characteristics associated with wetlands. Such areas occurring within floodplains are more appropriately termed nonwetlands.
WATERSHED	The region drained by or contributing water to a stream, lake, or other body of water.
WATER TABLE	The upper surface of groundwater or that level below which the soil is saturated with water. It is at least 6 inches thick and persists in the soil for more than a few weeks.
WETLANDS	Those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.

SPEAKERS, PANELISTS, MODERATORS

Paul Adamus
Corvallis Environmental Research Lab
200 SW 35th St.
Corvallis, OR 97333
503-757-4666

Dr. Chris Athanas
Univ. of Maryland, Horn Point
PO Box 775
Cambridge, MD 21613
301-530-5979

Dr. Michal J. Bardecki
Ryerson Polytechnical Institute
350 Victoria St.
Toronto, CANADA M5B 2K3
416-979-5038

Thomas W. Bilhorn
Earth Sciences Consultants
17456 Drayton Hall
San Diego, CA 92128
619-485-6457

Stephen J. Brady
USDA, Soil Conservation Service
3825 Mulberry
Ft. Collins, CO 80525
303-224-1885

Dr. Robert P. Brooks
Pennsylvania State University
Forest Resource Laboratory
University Park, PA 16802
814-863-1596

Dr. Robbin Brown
US Geological Survey
702 Post Office Bldg.
St. Paul, MN 55101
612-725-7841

Larry Burns
U.S. EPA, Athens Lab
College Station Rd.
Athens, GA 30613
404-546-3134

Virginia Carter
US Geologic Survey
Mail Stop 431
12201 Sunrise Valley Drive
Reston, VA 22092
703-648-5897

Dr. Ellis J. Clairain
US Army Corps of Engineers
PO Box 631
Vicksburg, MS 39180
601-634-3774

Michael Duever
National Audubon Society
Box 1877 Route 6 Sanctuary Rd.
Naples, FL 33964
813-657-2531

Douglas Ehorn
US EPA
230 South Dearborn
Chicago, IL 60604
312-886-0139

John Elder
US Geological Survey
6417 Normandy Lane
Madison, WI 53719
608-274-3535

Dr. Kevin L. Erwin
Consulting Ecologist
2077 Bayside Parkway
Fort Myers, FL 33901
813-337-1505

Jeanette Fitzwilliams
13 W. Maple Street
Alexandria, VA 22301
703-548-7490

Ronald Flanagan
R.D. Flanagan and Associates
201 West 5th St. Suite 140
Tulsa, OK 74103
918-587-7166

Dr. Mark S. Fonseca
Southeast Fisheries Center
Beaufort Laboratory
Beaufort, NC 28516-9722
FTS 670-9729

Dr. Edgar W. Garbisch, Jr.
Environmental Concern, Inc.
PO Box P
St. Michaels, MD 21663
301-745-9620

Dr. J. Wendall Gilliam
North Carolia State University
Soil Sciences, Box 7619
Raleigh, NC 27695
919-556-4860

Jeffrey Haltiner
Philip B. Williams & Assoc.
Pier 33 N., The Embarcadero
San Francisco, CA 94111
415-981-8363

Richard Hamann, Esq.
Center for Governmental Responsibility
230 Bruton-Geer Hall
Gainesville, FL 32611
904-392-2237

Judson W. Harvey
University of Virginia
Dept. of Environmental Sciences
Charlottesville, VA 22901
804-924-0560

Scott Hausmann
WI DNR
PO Box 7921
Madison, WI 53707
608-266-7360

Dr. Donald Hey
Wetlands Research, Inc.
53 Jackson Blvd., Suite 1401
Chicago, IL 60604
312-922-0777

Dr. Daniel Hubbard
South Dakota State University
Wildlife & Fisheries Science
Brookings, SD 57007
605-688-6121

Marvin Hubbell
Illinois Dept. of Conservation
524 S. 2nd St.
Springfield, IL 62701
217-782-3715

Dr. Terry Huffman
Huffman Technologies
69 Aztec St.
San Francisco, CA 94110
415-821-4159

Martha Jarosewich
WAPORA
1555 Wilson Boulevard,
Suite 700
Rosslyn, VA 22207
703-524-1171

Larry R. Johnson
LR Johnson Associates
31 Franklin Street
Westport, CT 06880
203-226-9383

Peggy B. Johnson
Clinton River Watershed
8215 Hall Road
Utica, MI 48087
313-739-1122

Dr. Robert G. Kadlec
Dept. Chemical Engineering
3094 Dow Bldg.
University of Michigan
Ann Arbor, MI 48109
313-764-3362

Mary Kentula
Northrop Services, Inc.
200 SW 35th Street
Corvallis, OR 97330
503-753-6221

Charles V. Klimas
US Army Corps of Engineers
Waterways Experiment Station
Vicksburg, MS 39180-0631
601-634-2983

Mark L. Kraus
Hackensack Meadowlands D.C.
One DeKorte Park Plaza
Lyndhurst, NJ 07071
201-460-1700

David F. Lakatos
Roy F. Weston, Inc.
Weston Way
West Chester, PA 19380
215-692-3030

Mary C. Landin
US Army Corps of Engineers
Waterways Experiment Station
Vicksburg, MS 39180
601-634-2983

Dr. Edward T. LaRoe
US Fish & Wildlife Service
Matomic Bldg., Room 555
Washington, DC 20240
202-653-8723

Larry Larson
WI DNR
PO Box 7981
Madison, WI 53707
608-266-1926

Steven Leimer
Envirodyne Engineers
168 North Clinton Street
Chicago, IL 60606
312-648-1700

Daniel A. Levine
School of Public and Environmental Affairs
Indiana University
Bloomington, IN 47405
812-332-9485

Dr. R. Roy Lewis
Housel & Associates
PO Box 82066
16105 N. Florida Ave.
Tampa, FL 33682
813-961-1444

Dr. Orie L. Loucks
Holcomb Research Institute
Butler University
Indianapolis, IN 46208
317-283-9667

Allan M. Lumb
USGS-WRD MS 415
12201 Sunrise Valley Drive
Reston, VA 22092
703-648-5306

Keith B. Macdonald
Keith B. Macdonald & Associates
PO Box 60310
San Diego, CA 92106
619-224-6643

Michael Mantell, Esq.
The Conservation Foundation
1250 Twenty-Fourth St. N.W.
Washington, DC 20037
202-293-4800

Lauressa McNemar
WB Satterthwaite Assoc.
720 North Five Point Rd.
Westchester, PA 19380
215-692-5770

Bradley Miller
U.S. EPA
999 18th St.
Denver, CO 80302
303-293-1570

Chris Miller
Greenhorne and O'Mara, Inc.
9001 Edmonston Rd.
Greenbelt, MD 20770
301-982-2000

Pat Nelson
US Fish & Wildlife Service
2627 Redwing Road
Fort Collins, CO 80526-2899
303-226-9379/323-5397

Dr. William A. Niering
Biology Department
Connecticut College
New London, CT 06320
203-447-1911

Dr. Richard Novitzki
US Geological Survey
521 W. Seneca St.
Ithaca, NY 14850
607-272-8722

William K. Nuttle
Dept. of Environmental Sciences
University of Virginia
Charlottesville, VA 22903
804-924-7761

Dr. Arnold O'Brien
University of Lowell
One University Ave.
Lowell, MA 01854
617-452-5000

Robert J. O'Brien
Inter-Fluve, Inc.
13111 Alamea Parkway, Box 28147
Denver, CO 80228
406-586-8926

Gene Parker
US Geological Survey
150 Causeway Street
Boston, MA 02114
617-223-6957

Colen Peters
Timson, Schepps & Peters, Inc.
103 Water St., Suite 202
Hallowell, ME 04347
207-623-0053

Donald L. Raisanen
Envirodyne Engineers, Inc.
168 North Clinton St.
Chicago, IL 60606
312-648-1700

Scott B. Robertson
ARCO Alaska, Inc.
PO Box 100360
Anchorage, AK 99510
907-276-1215

Dr. J. Henry Sather
US Fish & Wildlife Service
103 Oakland Lane
Macomb, IL 61455
309-833-5341

Dr. Joseph Shisler
Environmental Connection
P.O Box 69
Prerrinville, NJ 08535
201-446-3669

William Sipple
US EPA
401 M Street
Washington, DC 20460
202-382-5066

Wayne Skaggs
North Carolina State University
Soil Sciences, Box 7619
Raleigh, NC 27695
919-556-4860

James R. Stark
US Geological Survey
702 Post Office Building
St. Paul, MN 55101
612-725-7841

Nancy Suurballe
US Geological Survey
150 Causeway St., Suite 1001
Boston, MA 02114
617-223-6957

Nancy C. Taylor
US EPA
849 Chestnut Bldg. 3ES42
Philadelphia, PA 19107
215-597-6289

Russell Theriot
US Army Corps of Engineers
Waterways Experiment Station
CEWES-ER, P.O. Box 631
Vicksburg, MS 39180
601-634-2733

G. Nicholas Textor
Envirodyne Engineers
168 North Clinton Street
Chicago, IL 60606
312-648-1700

Dr. Frank Thomas
FEMA
500 C St., S.W.
Washington, DC 20472
202-646-2717

Edward Thomas, Esq.
FEMA, Region 1
J.W. McCormack Bldg.
Boston, MA 02109
617-223-9561

Richard Twait
Illinois Water Survey, Water Quality
Box 697
Peoria, IL 61652
309-671-3196

Harold A. Vance
J.M. Tettemer & Associates
1952 Fairburn Avenue
Los Angeles, CA 90025-5912
213-474-8338

Robert Watson
Wisconsin DNR
Box 7981
Madison, WI 53707
608-267-9868

Dr. Milton Weller
Dept. of Fisheries & Wildlife
Texas A & M University
College Station, TX 77843
409-845-5777

Dr. Daniel E. Willard
School of Public & Env. Affairs
Indiana University
Bloomington, IN 47405
812-332-9485

Dr. Philip Williams
Philip Williams & Associates
Pier 33 North, The Embarcadero
San Francisco, CA 94111
415-981-8363

Brian H. Winchester
Winchester Environmental Associates
7325 Northeast 13th Blvd., Suite 88
Gainesville, FL 32606
904-376-6500

Dr. Joy B. Zedler
Dept. of Biology
San Diego State University
San Diego, CA 92182-0057
619-265-5809

Dr. James Hall Zimmerman
University of Wisconsin
25 Agricultural Hall
Madison, WI 53706
608-423-4047